

# NITRIFICATION IN ACTIVATED SLUDGE PLANTS GUIDELINES ON SOME OPERATION AND DESIGN ASPECTS

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NITRIFICATION IN ACTIVATED SLUDGE PLANTS  
GUIDELINES ON SOME OPERATION AND DESIGN ASPECTS

By:

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Research Publication W62

Wastewater Treatment Section  
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## 1. SUMMARY

Since an increasing number of wastewater treatment facilities are being required to achieve nitrogenous oxidation (nitrification) as well as carbonaceous removal, a review of the current knowledge for the attainment of nitrification in an activated sludge plant was undertaken.

Information gained from a literature search and from the Ministry of the Environment's nitrification-denitrification studies<sup>(32)</sup> is reviewed.

The factors that affect nitrification in the activated sludge process are discussed and optimum values of process parameters are suggested for design and operation.

The importance of, and the techniques involved in, wastewater treatability studies to aid in nitrification plant design are discussed and process monitoring requirements are outlined.

## 2. INTRODUCTION

A considerable number of existing activated sludge plants designed for the removal of carbonaceous Biochemical Oxygen Demand ( $BOD_5$ ), allow the discharge of ammonia-nitrogen to the receiving waters.

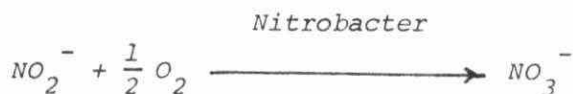
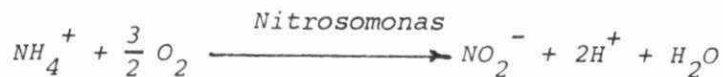
This ammonia-nitrogen i) exerts a Nitrogenous Oxygen Demand (NOD) on the receiving waters, ii) is toxic in an un-ionized form ( $NH_3$ ) to aquatic life, iii) adds to downstream eutrophication and iv) prevents superior disinfection by free chlorine (as  $HOCl$ ) from being exerted. (Additional information on the effects of ammonia concentration on disinfection is given in Appendix 12.5).

Due to these adverse effects, regulatory agencies in Ontario, along with other groups, are now recommending that plant effluents contain less nitrogenous as well as carbonaceous material (1). The Water Pollution Control Plant (WPCP) effluent criteria can be stated in terms of allowable ammonia ( $NH_3-N$ ), NOD and/or Total Oxygen Demand (TOD)\* in either mg/l or lbs/day discharge to a given water body.

The oxygen demand delivered to a receiving water is more critical during the summer months. Consequently, the option for separate summer and winter WPCP discharge requirements is now being considered by the Ministry of the Environment.

Studies indicate that it is possible to achieve a high degree of nitrification in the activated sludge process by propagating the nitrogenous oxidation reactions that normally occur in the surface waters. These reactions involve the oxidation of ammonia to nitrites and then to nitrates. This conversion is accomplished biologically mainly by the autotrophic bacteria *Nitrosomonas* and *Nitrobacter* as follows: (Painter (22))

\* The Total Oxygen Demand represents the combined carbonaceous and nitrogenous oxygen demands.



It is further possible, if required, to reduce the nitrates to nitrogen gas in a denitrification basin and hence eliminate the nitrogen compounds completely. However, the denitrification process does not form a part of this review.

## 2.1 Nitrification in Ministry of the Environment WPCP's

Results from the Ministry of the Environment's Plant Operations Summaries for 1973, 1974 and 1975 show that between 18 and 44% of the conventionally designed plants achieve a mean effluent ammonia nitrogen below 3 mg/l (3).

Extended aeration facilities produced somewhat better results with 44 to 89% of the plants giving mean effluent values less than 3 mg/l ammonia-nitrogen for the same period.

The reasons for a lack of full nitrification (i.e. complete oxidation of ammonia compounds) in the majority of the MOE plants are not completely clear from the available data. However, the Ministry's technical advisory work at many activated sludge plants in Ontario indicate that inadequate air supply and/or lack of protection from erratic flows or from toxic waste are possible causes.

### 3. OBJECTIVES

The purpose of this report is to examine the process parameters affecting the nitrification process as applied to activated sludge plants, and to illustrate the use of these factors for facility design, evaluation and operation purposes.

#### 4. NITRIFICATION PROCESS CONFIGURATIONS

A variety of process configurations have been proposed and tested to attain a high degree of nitrification (4, 5). The following section delineates common system arrangements with the main differences being in the secondary treatment stage. These process variations are illustrated in Figure 1.

##### 4.1 Single-Stage Combined Sludge (S.S.C.S.)

The arrangement is similar to the conventional activated sludge plant design. To achieve a high degree of nitrification, the Solids Retention Time (SRT) is usually longer than that of conventional plants and a sludge recycle rate of at least 80% of influent flow is employed. BOD removal and ammonia oxidation take place simultaneously in one basin.

##### 4.2 Two-Stage Combined Sludge (T.S.C.S.)

This system incorporates two aeration basins in series, followed by clarification and sludge return as in the single-stage system. By splitting the aeration section, conditions more approximating plug flow are obtained.

##### 4.3 Two-Stage Separate Sludge (T.S.S.S.)

Two separate aeration basins are utilized with clarification and sludge recycle after each basin. BOD removal is attained in the first stage which operates as a high rate system with profuse sludge growth and wasting. The second stage functions as a low growth section for nitrification with limited wasting to maintain a nitrifying bacterial population.

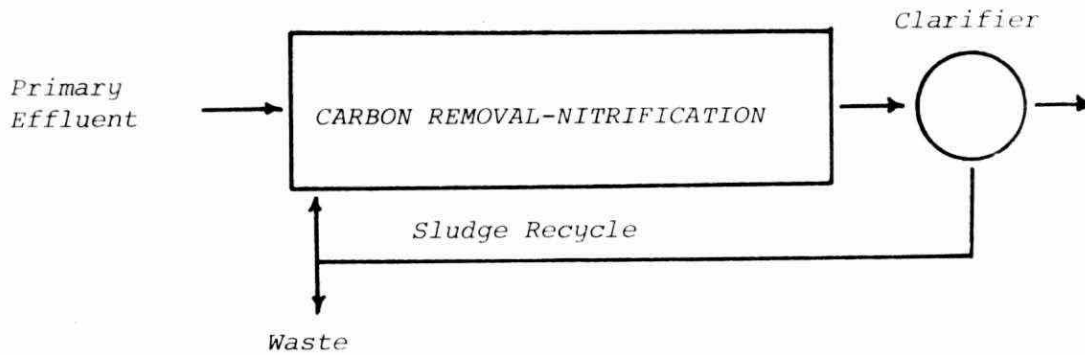


FIGURE 1

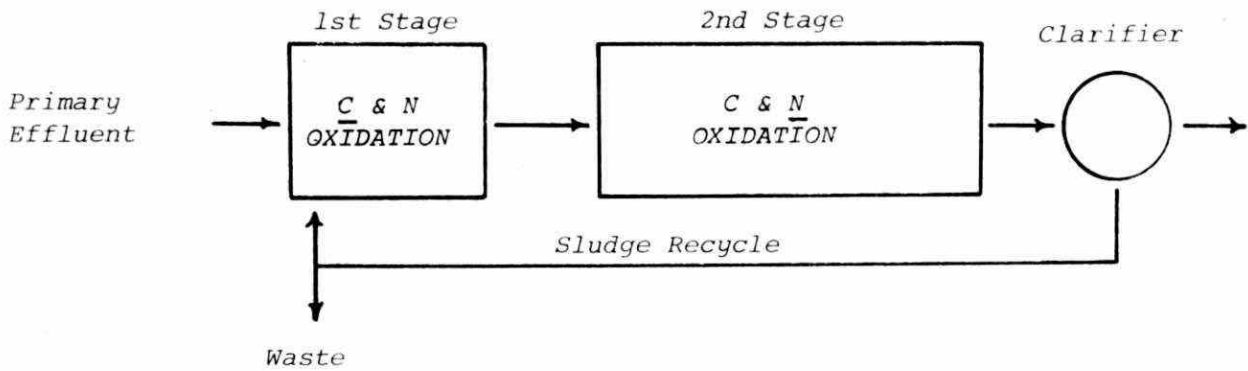
COMBINED AND SEPARATE CARBON REMOVAL

NITRIFICATION SLUDGE SYSTEMS

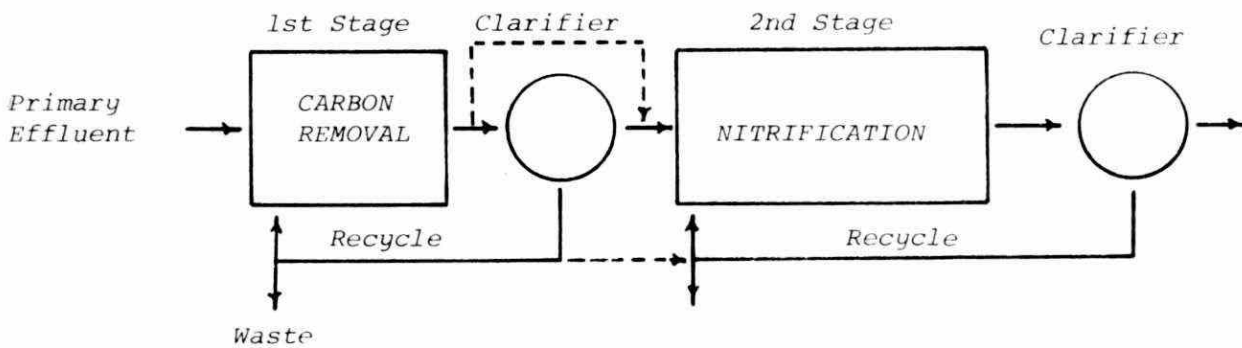
a) Single-Stage Combined Sludge



b) Two-Stage Combined Sludge



c) Two-Stage Separate Sludge



Some bypassing of primary effluent and/or first stage waste sludge to the second stage aeration basin is often recommended during periods of low sewage strength to maintain solids levels in the nitrification basin.

#### 4.4 Extended Aeration

Although not illustrated separately in Figure 1, the extended aeration process, similar to single-stage combined, is another system which is capable of achieving a high level of nitrification. This process can be designed as a completely mixed or as a plug-flow system such as an oxidation ditch. Because of a combination of long aeration detention times (18-24 hours) and low organic loadings, complete oxidation of ammonia compounds occurs and nitrification is maintained at low temperatures.

#### 4.5 Process Comparisons

In evaluating the relative merits or disadvantages of the above process configurations, factors such as reliability and ease of operation and capital cost should be considered.

The single-stage, combined-sludge system for nitrification is merely an adaptation of the conventional activated sludge design for BOD removal. Consequently, the modifications needed to form a nitrification facility are minimal and little or no extra man-hours are required for operation. The disadvantages of this arrangement are that because carbon removal and nitrification are accomplished in the same basin, less protection is afforded to the nitrification bacteria from the effects of toxic wastes.

The two-stage, combined-sludge system has the benefit of simplicity as with the single-stage system, but also has the separate first-stage basin which may help in buffering the effects of toxic waste prior to its entering the nitrification basin. The capital cost of this process is comparable to the single-stage system because the aeration requirements are similar.

The two-stage, separate-sludge configuration being somewhat more complex is more expensive and requires more operator attention. This system has the advantages of first stage buffering of toxic waste and of providing a finer control of the individual carbon and nitrogen conversion reactions. Because the wasting of sludge can be handled separately in the nitrification stage, process Solids Retention Time (SRT) can be controlled more readily.

The extended aeration process, having a long aeration period and SRT, is normally more stable, less affected by toxic wastes and generally requires less operator attention. Adaptability to a denitrification system is easier and less expensive than the aforementioned arrangement. The major drawbacks to extended aeration systems are the large land and oxygen requirements since a primary clarifier is not provided. (Refer to Section 5.3.2)

## 5. NITRIFICATION PROCESS PARAMETERS

A number of variables need to be optimized in order to ensure consistent operation of the nitrification process. Some of these factors depend on the waste flow quantity and quality, while others deal with the process operation.

### 5.1 Wastewater Characteristics

Proper design of the nitrification process requires a good knowledge of the daily flows and the physical and chemical characteristics of the wastewater. As many constituents (mainly from industrial sources) are toxic to the nitrifying bacteria (1, 6), it is imperative to control the level of toxicants.

#### 5.1.1 Loading

Raw sewage ammonia-nitrogen concentrations often vary by 60% of the mean daily level and do so in phase with flow fluctuations. The plotted data in Figure 2 illustrates a typical process nitrogen loading pattern. The peak of the flows and of the nitrogen concentration usually occur at mid-day and the effluent ammonia values often rise significantly for several hours following this period (Figure 3). In order to buffer these peaks and to redistribute more loading to the normally low flow periods, efforts are sometimes made to provide large storage capacity in the plant and hence equalize the flows. Some aspects of this process of equalization are discussed in Appendix 12.1.

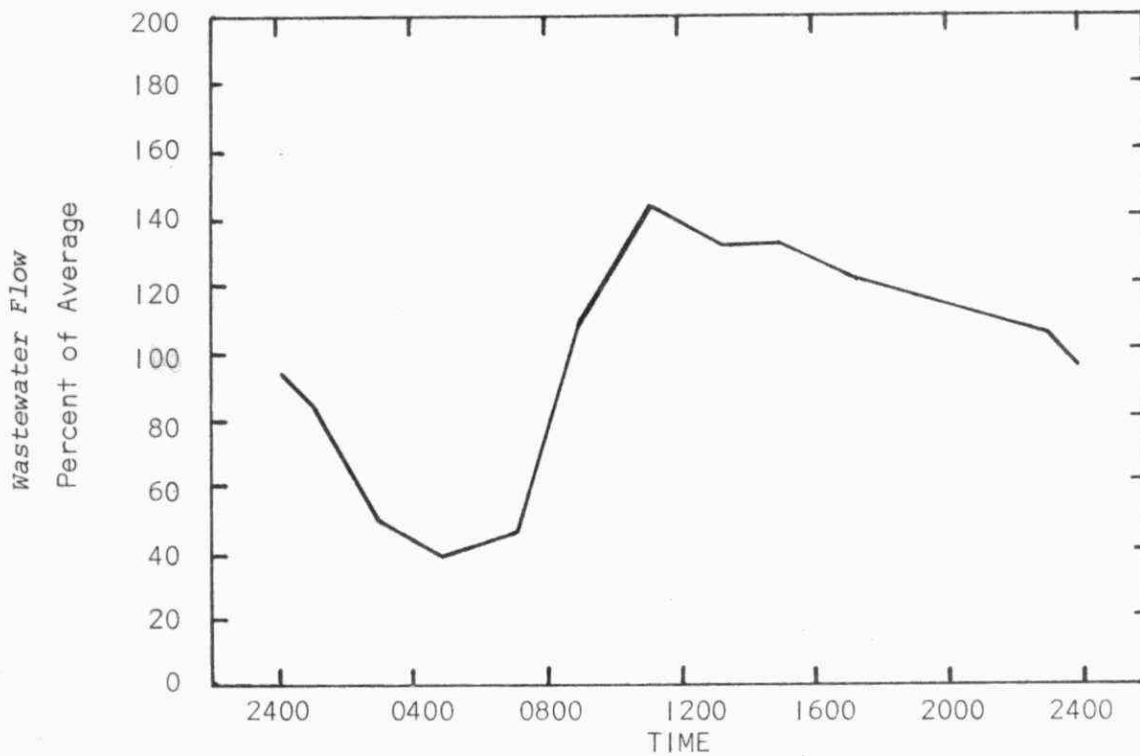
#### 5.1.2 Constituents

Normally, domestic sewage contains between 5-25 mg/l of ammonia-N.(3) Amounts in excess of this are usually a result of industrial dumping or possible septic conditions in the sewer system. The growth of nitrification bacteria such as *Nitrosomonas* and *Nitrobacter* is influenced by the concentration and form of available ammonia nitrogen.

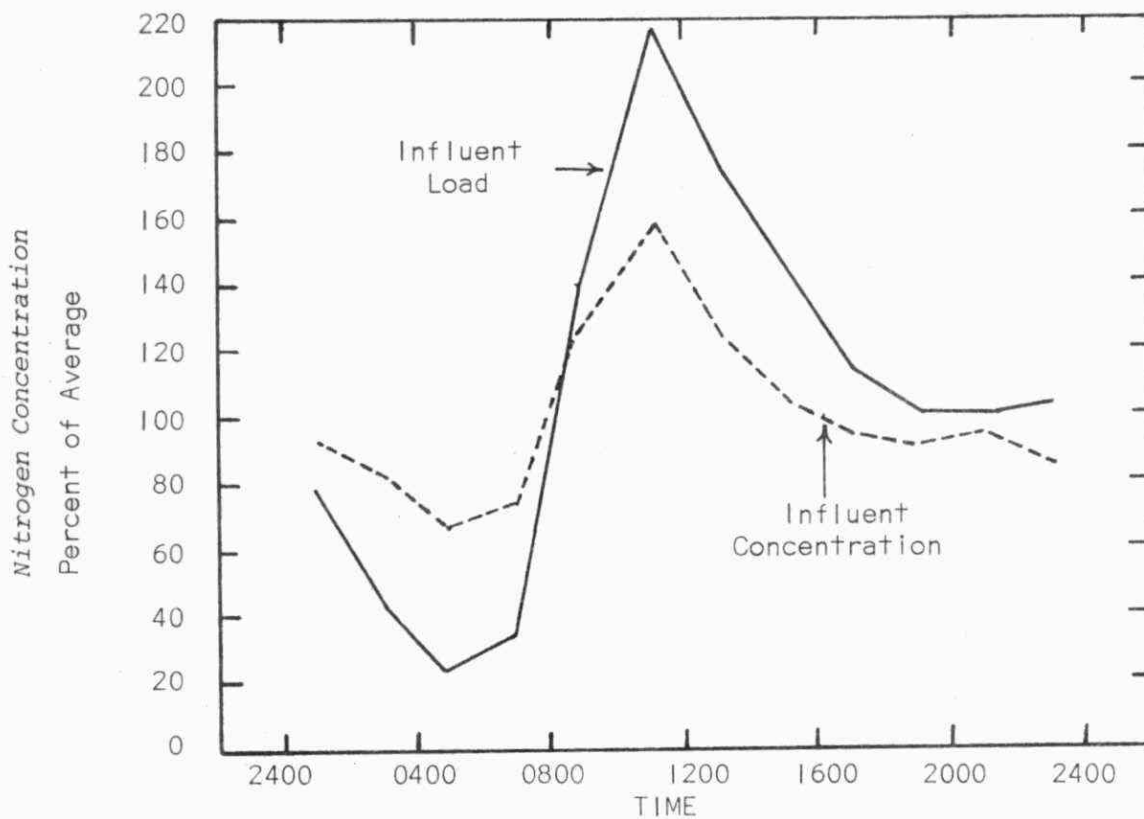
FIGURE 2

DIURNAL VARIATIONS AT THE CHAPEL HILL, N.C. TREATMENT PLANT

(USEPA 1975) (5)



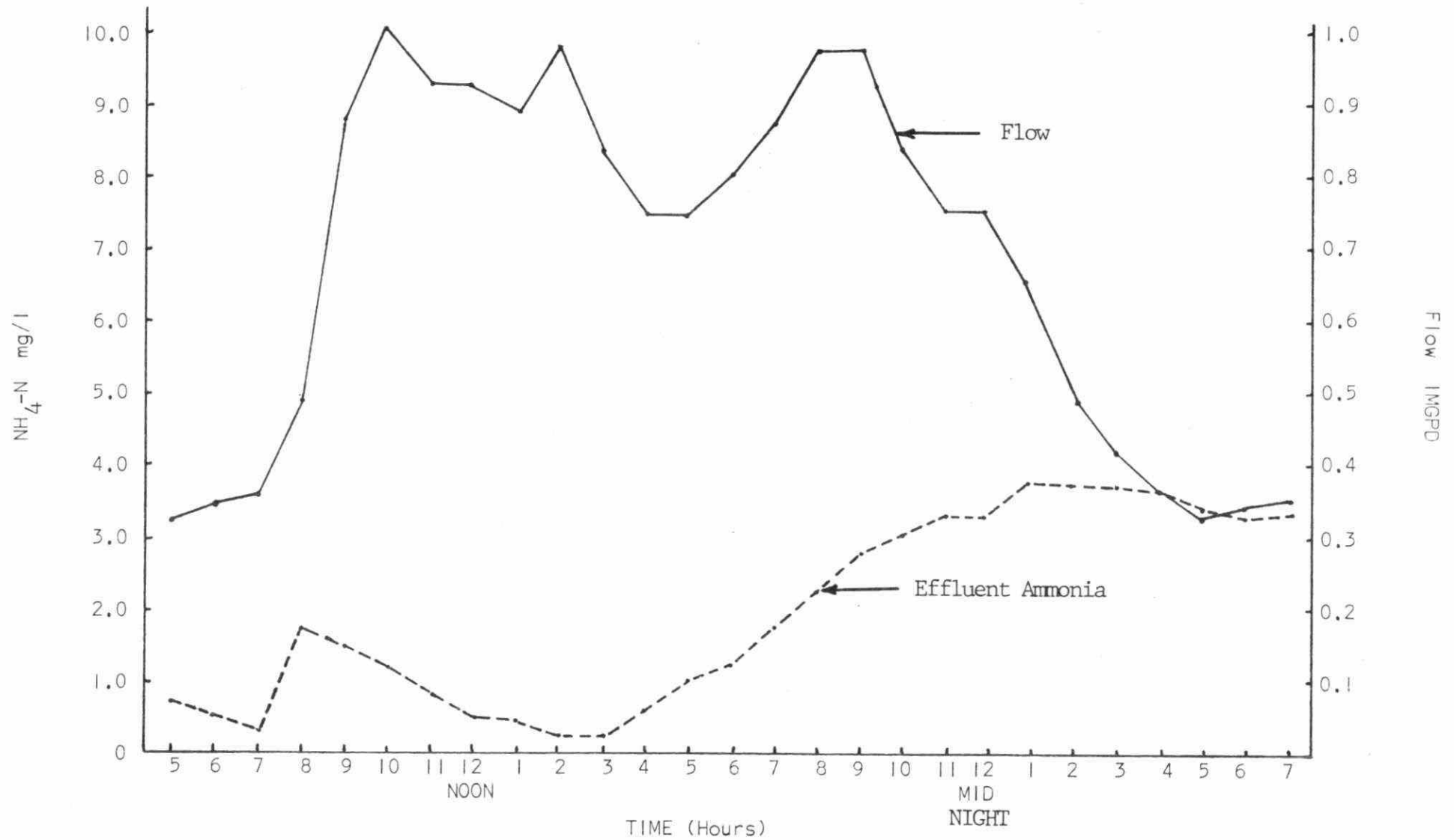
DIURNAL VARIATION IN WASTEWATER FLOW



DIURNAL VARIATION IN NITROGEN LOAD AND CONCENTRATION

FIGURE 3

EFFECT OF DIURNAL LOADING ON EFFLUENT AMMONIA CONCENTRATION  
(O.E.F. Nitrification-Denitrification Process) (12)



The toxicity of Free Ammonia (un-ionized) to nitrogen bacteria has been reported by Prakasam and Loehr (7) and Anthonisen et al (8). Inhibition of Nitrosomonas and Nitrobacter by Free Ammonia (FA) was found to begin at levels of 10 to 150 mg/l and 0.1 to 1.0 mg/l, respectively. As indicated in Figure 4, from Anthonisen's work the Free Ammonia content in sewage is a function of ammonia concentration and pH at constant temperature. As the levels of ammonia and pH rise in wastewater, so does the concentration of un-ionized ammonia.

Similar effects on these two nitrifying bacteria groups were also reported with Free Nitrous Acid (FNA) at concentrations of 0.2 mg/l and 2.8 mg/l on Nitrosomonas and Nitrobacter, respectively, depending also on wastewater pH (Figure 5). If nitrite is present, a conversion to FNA will occur upon lowering pH.

Wastewater temperature is also an important factor in the release of FA and FNA. Interpretation of data of Duisberg and Buehrer by Anthonisen et al (8) revealed that upon raising temperature from 20 to 30°C, at similar pH and ammonia concentration, the value of FA doubled.

Anthonisen et al, presented the following equations for the effect of ammonia concentration, pH and wastewater temperature on FA and FNA production:

$$(1) \quad \text{FA (as NH}_3\text{) (mg/l)} = \frac{17}{14} \times \frac{\text{total ammonia as N (mg/l)} \times 10 \text{ pH}}{K_b/K_w + 10 \text{ pH}}$$

where  $K_b$  is the ionization constant for ammonia and  $K_w$  is the ionization constant for water and is defined as:

$$(2) \quad K_b/K_w = e^{\left(\frac{6344}{273 + ^\circ\text{C}}\right)}$$

FIGURE 4

DIAGRAM FOR DETERMINING UN-IONIZED AMMONIA CONCENTRATION  
(After Anthonisen et al, 1974) (8)

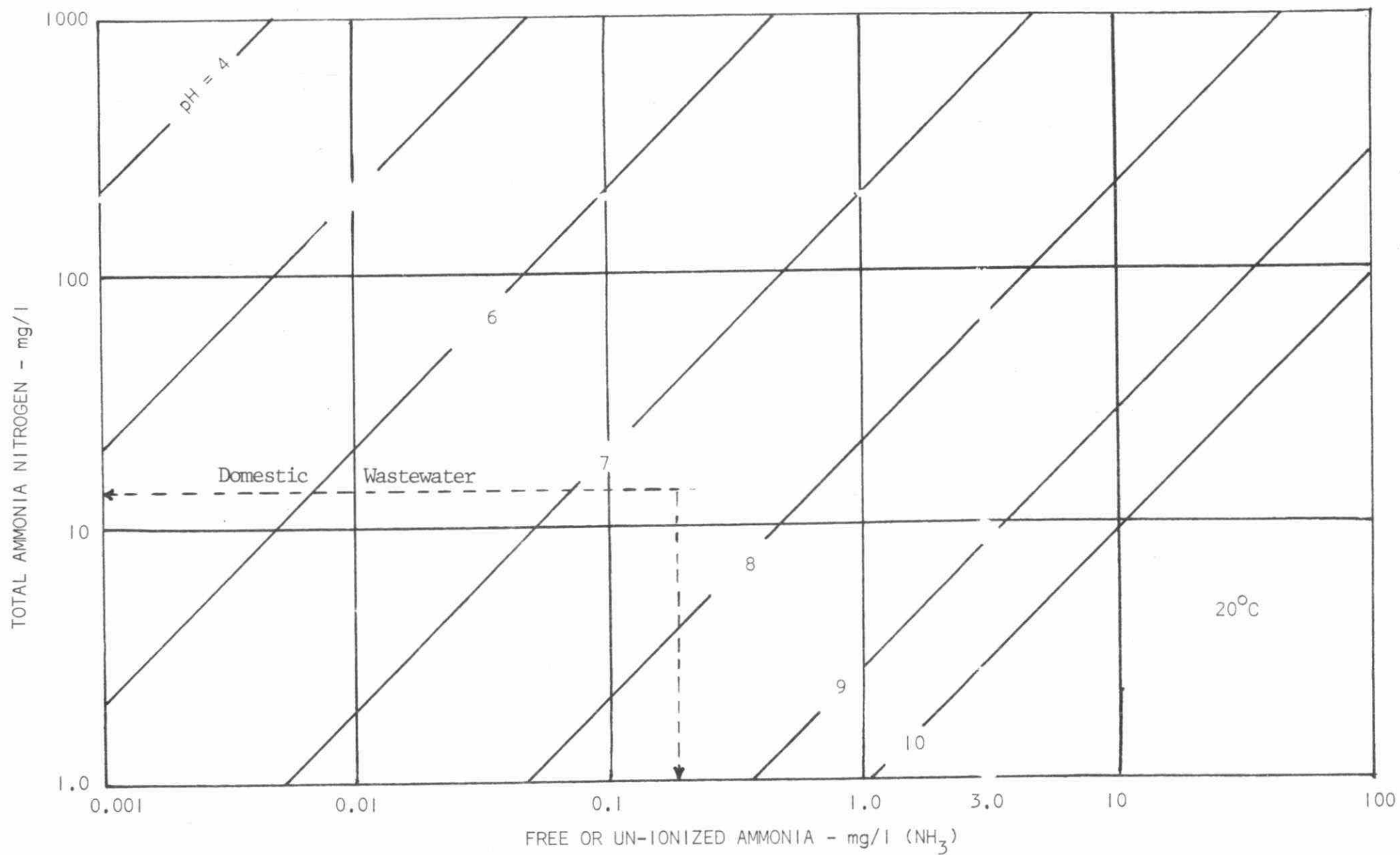
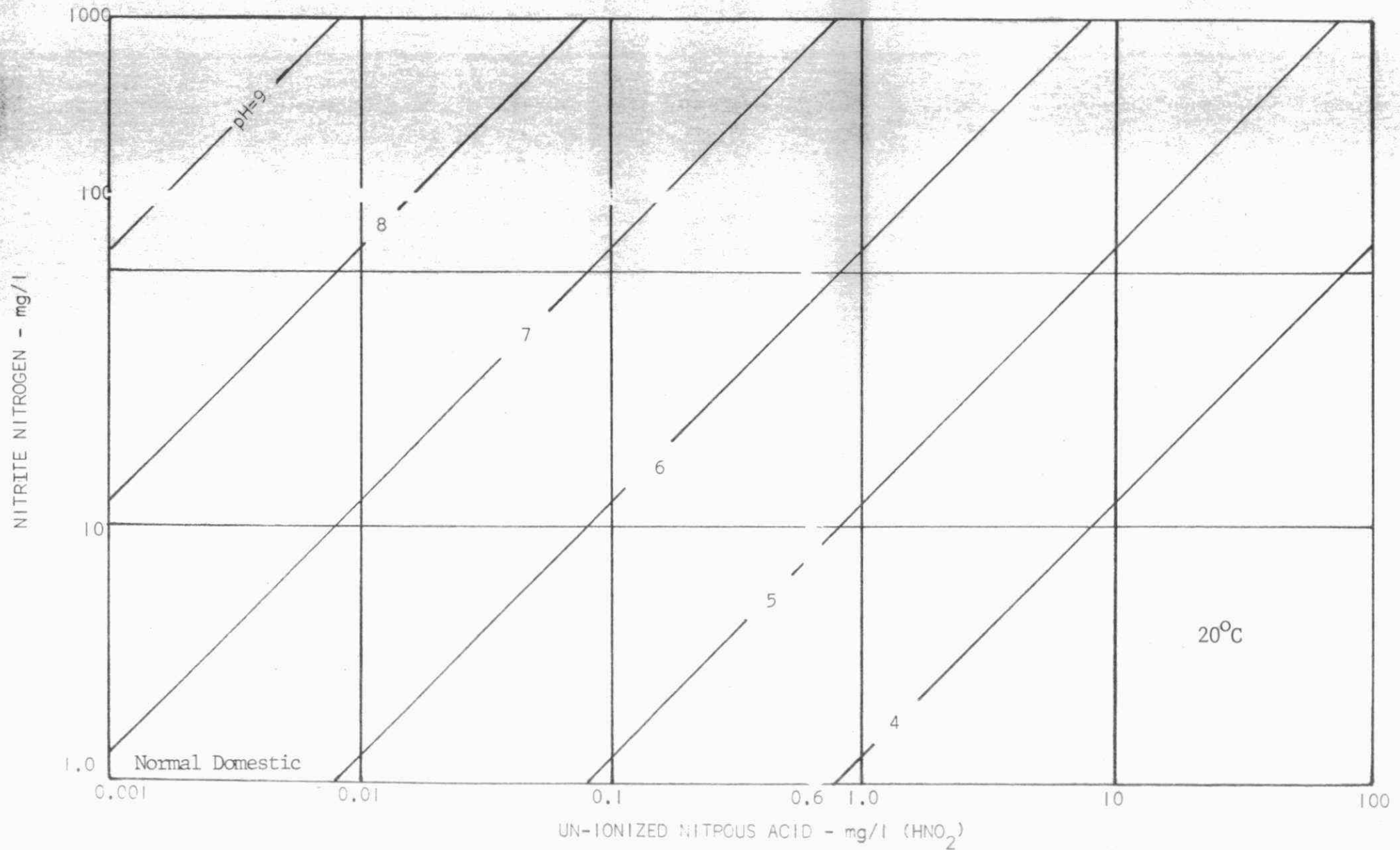




FIGURE 5

DIAGRAM FOR DETERMINING UN-IONIZED NITROUS ACID CONCENTRATIONS  
(After Anthonisen et al, 1974) (8)



$$(3) \text{ FNA (as HNO}_2\text{) (mg/l)} = \frac{46}{14} \times \frac{\text{NO}_2\text{ (as N) mg/l}}{K_a \times 10^{\text{pH}}}$$

where  $K_a$  is the ionization constant for nitrous acid

$$(4) K_a = e^{\left(\frac{-2300}{273 + ^\circ\text{C}}\right)}$$

Generally, domestic wastewater will not present constituents and conditions for the formation of FA and FNA as toxicants to nitrification bacteria. These substances are more apt to appear in industrial wastes having high ammonia concentrations (more than 30 mg/l as N) with pH's in the acidic <6.5 or basic range >8.5 and temperature above 20°C.

Considerable work has been done on the role of materials from industrial sources which are toxic to the nitrifying bacteria and as such, have a major influence on the design of the process. These substances are generally i) metallic compounds such as those of Cu, Na, Cr, Ni, etc., ii) organic compounds such as fungicides, accelerators and those in which the sulphur and nitrogen atoms are attached to the same C atom, and iii) oxidizing compounds such as chlorine and hydrogen peroxide used for treatment of bulking sludges. The details of the toxicity levels are presented in Appendix 12.2.

## 5.2 Primary Clarification

Full-scale process evaluations by the Ministry with nutrient removal facilities have shown that considerable carbonaceous and nitrogenous matter can be removed prior to secondary treatment by the addition of chemicals or enlargement of primary clarifier capacity. Studies by Black (9) at the Newmarket WPCP showed that a 30% greater BOD<sub>5</sub> reduction across the

primary clarifiers was attainable with lime addition to the raw sewage. Ammonia reduction was also seen frequently, probably due to stripping at high pH values (Table 1).

Similar observations were noted in recent studies by the author (10) at the Ontario Experimental Facility (OEF) on nitrification-denitrification studies. A primary clarifier having a detention time of 4 hours and a mean overflow rate of 340 gal/sq ft/day ( $17 \text{ m}^3/\text{m}^2/\text{d}$ ) achieved 63% reduction of  $\text{BOD}_5$  and an 80% reduction in suspended solids, compared to a 30 and 43% reduction in  $\text{BOD}_5$  and suspended solids respectively, in other MOE conventional plants.

Costs analyses of wastewater treatment plants by Smith (11) in 1968 produced evidence that the construction and maintenance costs and power requirements of primary units are less than those required for secondary facilities. This may not always be the case depending on the location, availability and costs of equipment, but enlargement or chemical treatment of the primary process should be considered.

### 5.3 Aeration Basin Parameters

The major operational parameters of the nitrification process are related to the aeration (nitrification) basin and these require careful monitoring.

#### 5.3.1 Alkalinity and pH

The biochemical reactions involved in the oxidation of ammonia and bacterial synthesis are as follows (5):

##### i) Synthesis - Oxidation

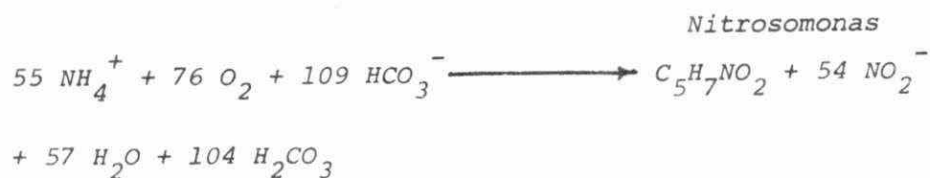


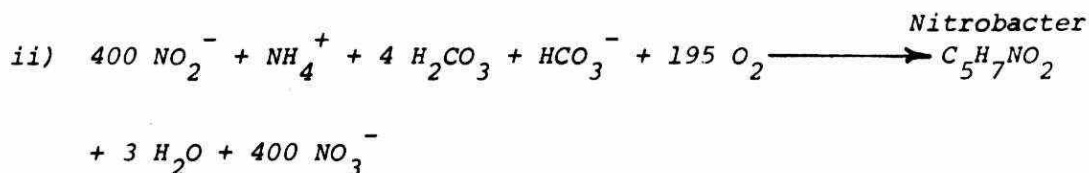
TABLE 1  
MOE LIME ADDITION STUDIES  
MEAN ANALYTIC RESULTS

Newmarket WPCP (Black, 1972) (9)

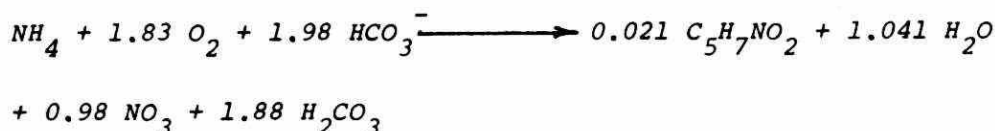
	<u>Before Lime Addition</u>							<u>After Lime Addition</u>						
	BOD	NH <sub>3</sub> -N	<u>Nitrogen</u> Tot. Kj. -N	NO <sub>3</sub> -N	Hardness	Alkal.	pH	BOD	NH <sub>3</sub> -N	<u>Nitrogen</u> Tot. Kj. -N	NO <sub>3</sub> -N	Hardness	Alkal.	pH
Raw Sewage Average	249	29	54	trace	225	386	7.8	227	28	47	trace	239	391	8.0
Prim. Effluent Average	172	27	46	trace	229	377	7.8	84	21	32	trace	240	374	9.6
Final Effluent Average	21	5.6	8.3	16.6	235	227	7.7	9	.8	2.9	19	277	247	8.0
Overall % Red.	92	81	85	-	-	41	-	96	97	95	-	-	37	-

Richmond Hill WPCP (Black and Lewandowski, 1969) (16)

Raw Sewage Average	126	18	33	0.2	412	387	-	139	18	33	0.12	401	381	-
Prim. Effluent Average	99	14	30	0.1	413	362	-	40	13	24	0.03	306	287	-
Final Effluent Average	53	15	22	0.6	413	342	-	5	.5	1.2	16	377	222	-
Overall % Red.	58	17	33	-	-	-	-	96	97	96	-	-	-	-



iii) Overall Synthesis and Oxidation



Oxidation reactions usually taking place at pH less than 8.3 involve the conversion of bicarbonates to carbonic acid. The carbonic acid is in turn utilized by the bacteria during synthesis as an aqueous source of carbon dioxide.

During these reactions, alkalinity ( $\text{HCO}_3^-$ ) is destroyed and acidity ( $\text{H}_2\text{CO}_3$ ) is produced. A theoretical value of 7.14 mg of alkalinity measured as  $\text{CaCO}_3$  is removed per mg of ammonia nitrogen oxidized (5) although actual experimental results (Table 2) show values ranging from 5.5 to 9.0.

The data in Figure 6 from nitrification-denitrification studies (12) at Kleinburg illustrate the removal of alkalinity with the onset of nitrification in the aeration tank. Information from these studies show an alkalinity loss of 7 to 8 mg as  $\text{CaCO}_3$  per mg ammonia nitrogen oxidized.

The addition of sodium hydroxide or lime to a nitrification process may be necessary in a locality of low alkalinity wastewater. This will prevent acidic conditions developing in the aeration basin due to ammonia oxidation reactions.

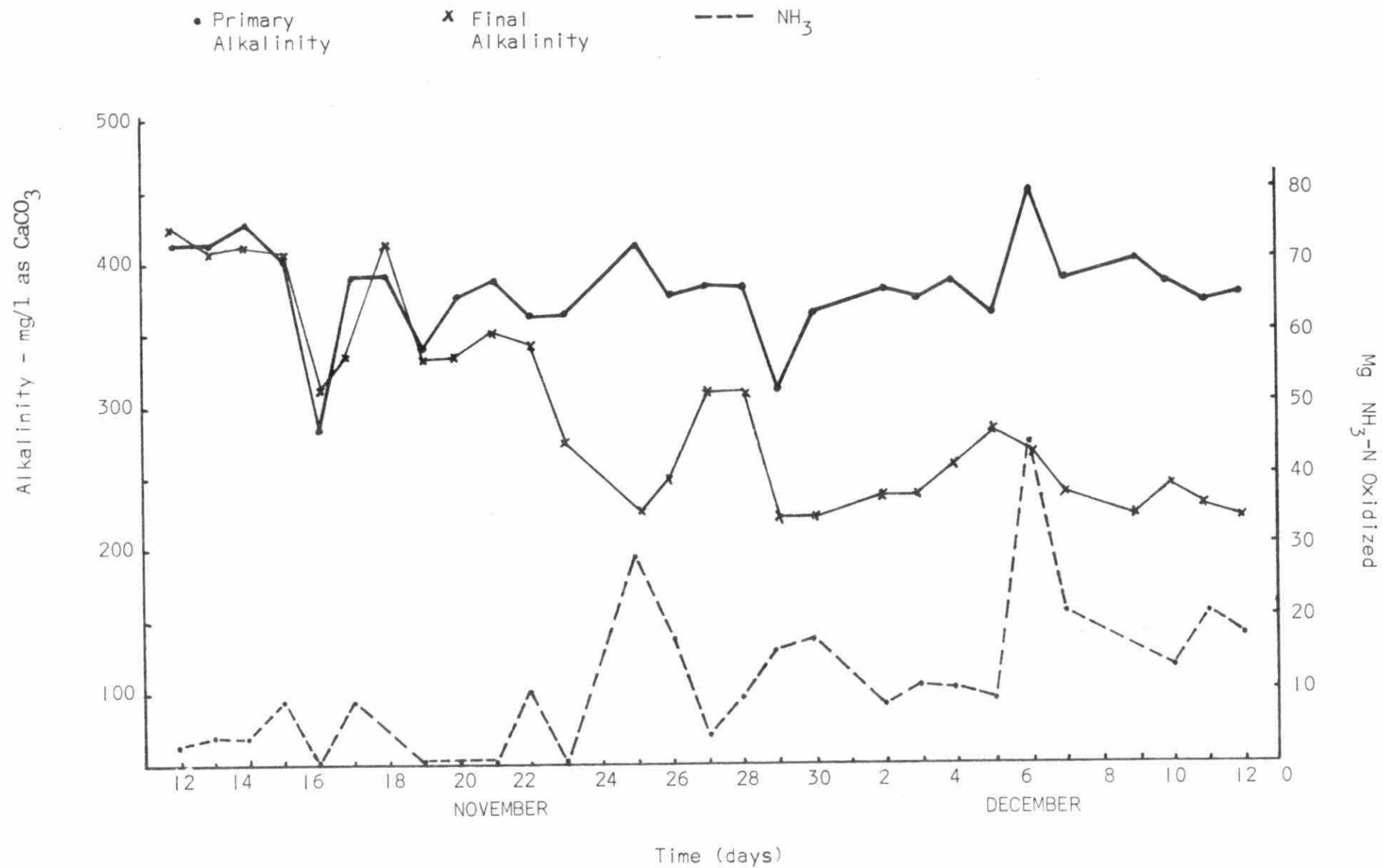
It has been generally reported (5) that upon raising aeration mixed liquor pH's from 7 to 9, a parallel increase in nitrification reaction rate will occur. Downing et al (13), however, stated that no stimulation in nitrification rate occurred in their experiments above pH 7.2 but an inhibition was evident below that value. Haug and McCarty suggested that acclimation by nitrogen bacteria to pH's lower than 7.0 may be possible.

TABLE 2  
ALKALINITY DESTRUCTION RATIOS IN EXPERIMENTAL STUDIES  
(U.S.E.P.A.) (5)

System	$\frac{\text{mg alkalinity destroyed}^a}{\text{mg NH}_4^+\text{-N oxidized}}$
Suspended growth	6.4
Suspended growth	6.0
Suspended growth	7.1
Attached growth	6.5
Attached growth	6.3 to 7.4
Attached growth	7.3

<sup>a</sup>As  $\text{CaCO}_3$ ; the theoretical value is 7.1

FIGURE 6  
ALKALINITY LOSS VS AERATION  $\text{NH}_3$  OXIDIZED  
(Smith) (12)



Rimer and Woodward (15) working with a two-stage pilot plant process found that by augmenting the aeration basin pH to 8.3 by the addition of sodium hydroxide, a marked increase in nitrification efficiency resulted.

An illustration of the influence of pH on nitrification rate is shown in Figure 7. The multi-plot from various investigations suggests an optimum range of between 8.3 and 8.5 for maximum nitrification.

Evidence of a pH effect on nitrification was shown during full-scale work by Black (9,16) (Table 1). Studies at the Richmond Hill and Newmarket WPCP's in which lime was added to raw sewage prior to primary and secondary treatment for phosphorus removal showed a marked increase in nitrification. In the case of the Newmarket plant (9), one of three aeration basins was taken out of service due to lowered organic loadings via first stage lime addition. The hydraulic detention time of the aeration basin was reduced from 10 to 6 hours and nitrification was maintained through the winter months. It was concluded that enhanced aeration pH levels (8 to 8.5) contributed to the consistent nitrification efficiency.

#### 5.3.2 Oxygen Requirements and Mixing

The level of aeration basin dissolved oxygen (DO) has been found to be a limiting factor for maintaining nitrification rate. As represented in Figure 8, studies by Nagal and Haworth on a single-sludge system show a definite relationship between the weight of ammonia oxidized per weight of MLSS (nitrification rate) and aeration DO concentration (17).

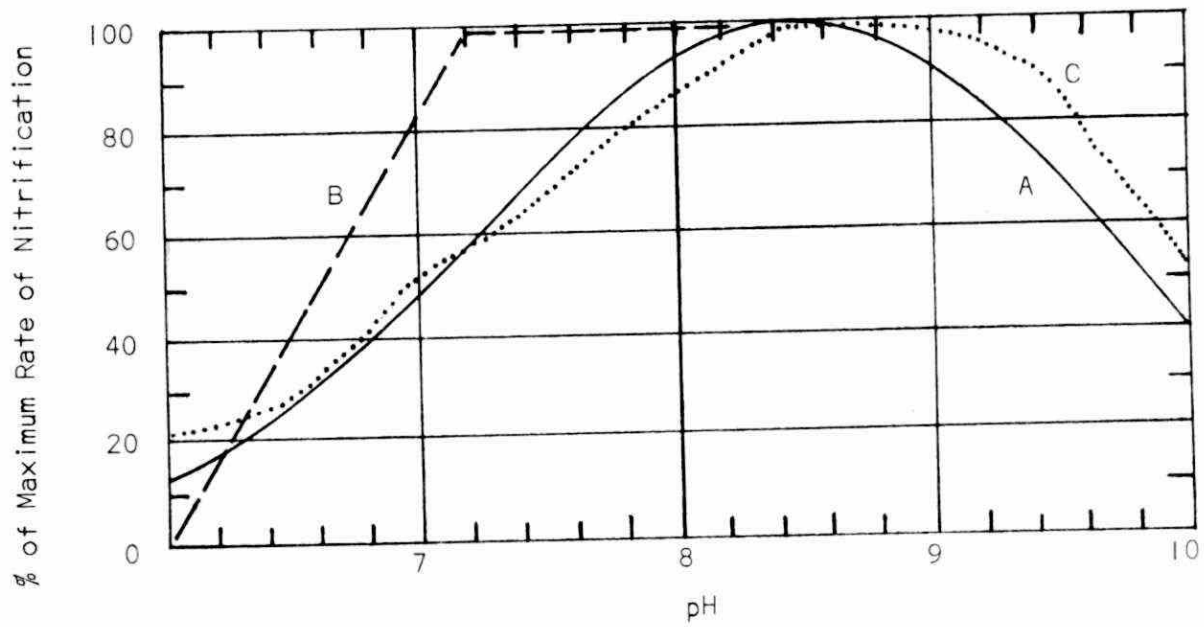
The United States Environmental Protection Agency (USEPA) (5) recommends a minimum aeration DO of 2 mg/l and studies by the author (12) on full-scale processes also indicate this to be the critical value.



FIGURE 7

EFFECT OF pH ON NITRIFICATION RATE

(U.S.E.P.A. Manual) (5)

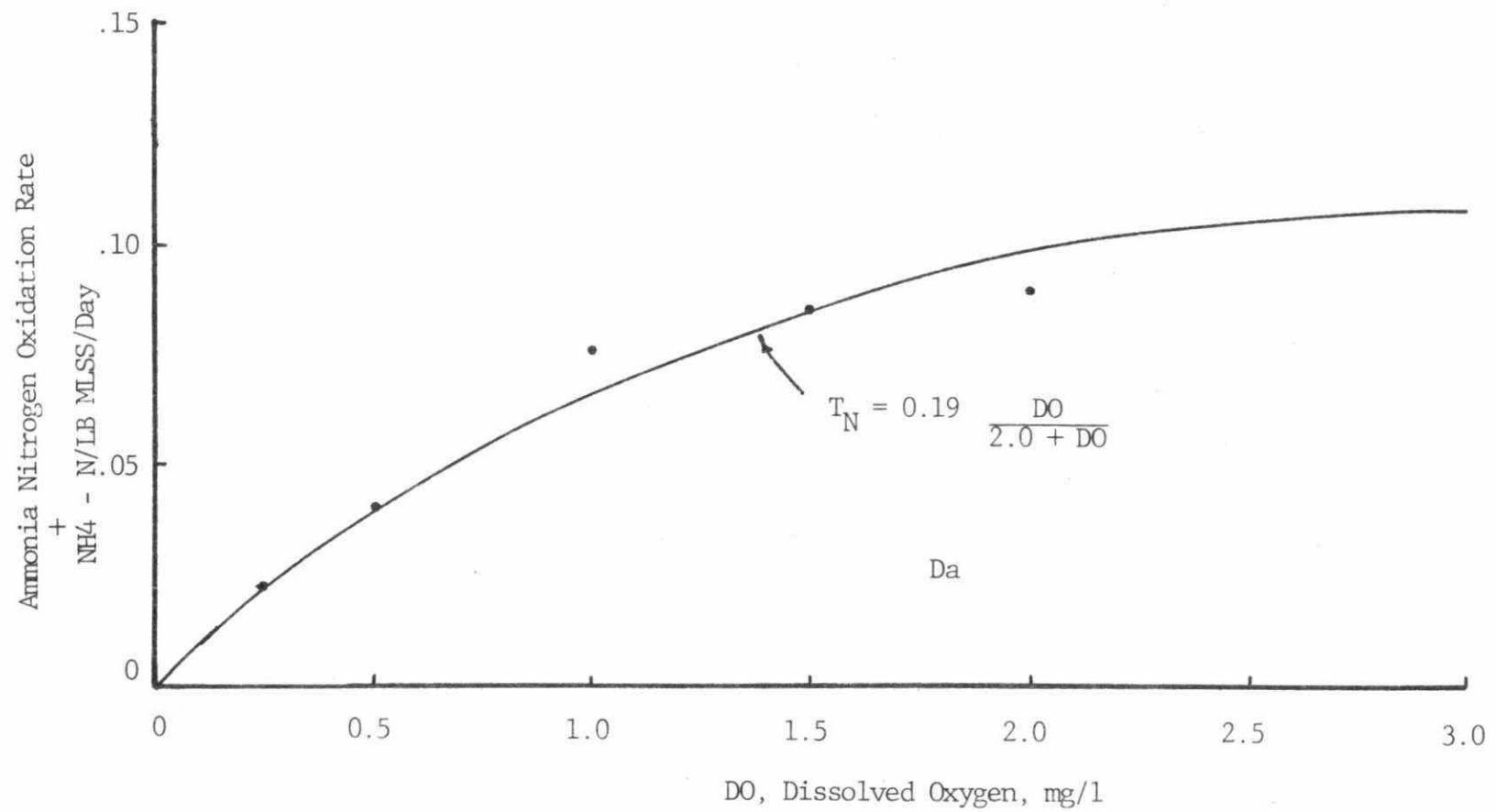


- A      activated sludge at 20°C
- B      activated sludge
- C      attached growth reactor at 20°C  
(i.e. Bio-contactor unit)

EFFECT OF DISSOLVED OXYGEN ON NITRIFICATION RATE

FIGURE 8

After Nagel and Haworth (17)



Aeration basins of longitudinal design giving close to plug flow conditions often have wide differences in DO along the length. This is caused by high organic loading to the initial stages of the aeration basin, particularly during peak flow periods. The DO levels can vary from 0.5 mg/l at the inlet, to 5.0 mg/l at the outlet as carbonaceous and nitrogenous matter is reduced along the length of the basin (12). This situation would cause uneven nitrification rate to occur in the basin with lower than optimal values produced in the first half. To combat uneven DO distribution, it is recommended that more oxygen and/or step-feeding of wastewater be available to the first half of a plug-flow basin (18).

From the standpoint of oxygen delivery to the activated sludge, a completely mixed system is superior. However, short-circuiting is more prevalent in "completely mixed systems", and effluent ammonia levels are usually higher than in plug-flow systems, particularly if flow-surfing is common. The square tank completely mixed system has been recommended for treatment of industrial wastes or domestic wastes with high percentage of industrial wastes (18, 5). Toxic materials on entering the completely mixed basin are diluted and evenly dispersed throughout and have less injurious effects upon the bacteria.

Conventional waste treatment plants designed for full nitrification require a substantially larger air capacity as 4.6 lbs of oxygen are required to fully oxidize 1 lb of ammonia nitrogen (5). To include carbonaceous demand, an extra 1.0 lbs oxygen per lb  $BOD_5$  applied (5) is also required.

Extended aeration plants (without primary sedimentation) treat a larger percentage of insoluble carbonaceous and nitrogenous material under conditions of long SRT (low F/M) (2). As a result of endogenous respiration, these materials are broken down and impose additional oxygen demand (49). Consequently, the oxygen requirements for extended aeration systems should be based on influent Total Kjeldahl Nitrogen ( $4.6 \text{ lbs } O_2/\text{lb TKN applied}$ ) for the nitrogenous fraction. To satisfy the carbonaceous oxygen demand,  $1.5 \text{ lbs } O_2/\text{lb BOD}_5$  applied is required (5).

It is possible to exercise more efficient control on the oxygen supply by an automatic instrumentated system. Details of this are given in Appendix 12.3.

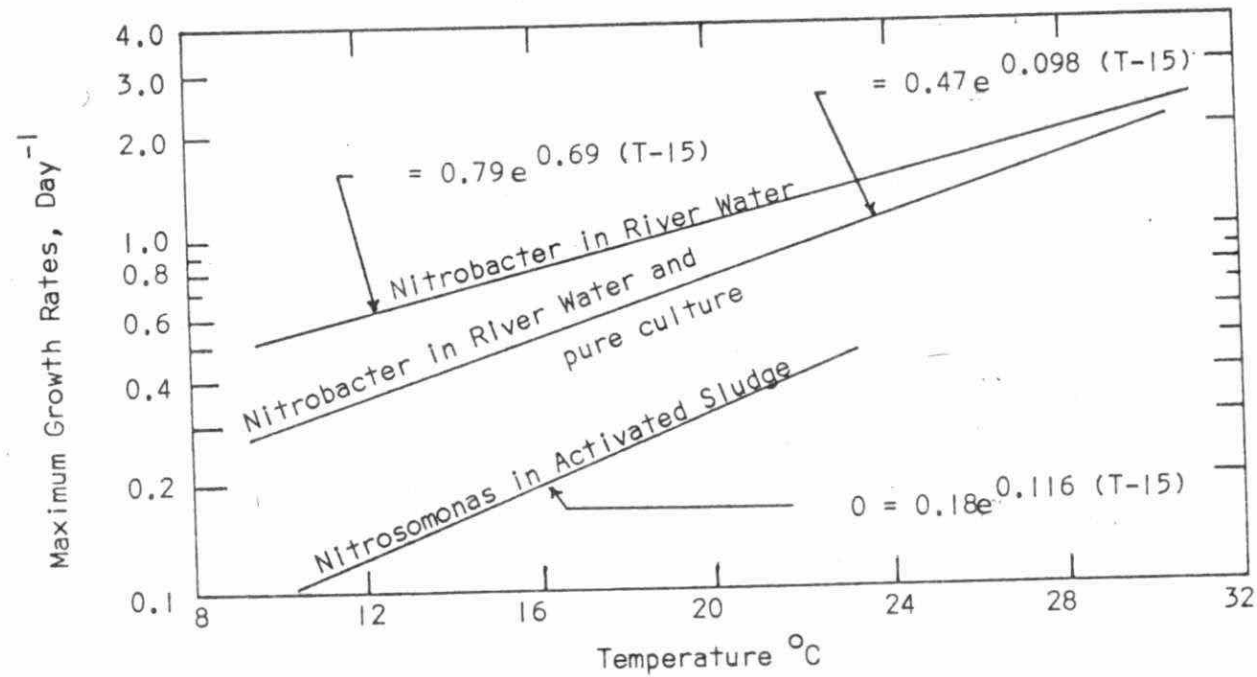
#### 5.3.3 Wastewater Temperature

A definite correlation can be shown between the relative growth rate of nitrification bacteria and the temperature of the wastewater medium. The growth rate of *Nitrobacter* although limited by the concentration of nitrite is considerably higher than *Nitrosomonas* which are dependent on ammonia nitrogen (19). As shown in Figure 9, both genera show an increase in growth rate with rising temperature. *Nitrosomonas* are the limiting microorganisms for nitrification because of their lower growth rate, consequently, their kinetics are used for design purposes.

Expressions for the effect of temperature ( $T^{\circ}\text{C}$ ) on maximum growth rate per day ( $\mu_N$ ) of *Nitrosomonas* have been presented by Downing et al (19) as follows:

FIGURE 9

TEMPERATURES DEPENDENCE OF THE MAXIMUM GROWTH RATES OF NITRIFIERS  
(U.S.E.P.A. MANUAL) (5)



$$(1) \hat{\mu}_N = 0.18e^{0.116 (T-15)}, \text{ for activated sludge}$$

$$(2) \hat{\mu}_N = 0.47e^{0.098 (T-15)}, \text{ for pure cultures}$$

Investigations by others such as Poduska and Andrews (20) and Lawrence and Brown (21) have shown values related closer to expression (2). Downing's results for activated sludge may have been limited by other parameters (5).

Expressions derived out of work on activated sludge by Knowles et al (5) are as follows: (temperature (T) range of 8 to 20°C)

$$(3) \log_{10} \hat{\mu}_N = 0.0413T - 0.944, \text{ for Nitrosomonas}$$

and

$$(4) \log_{10} \hat{\mu}_{NB} = 0.0255T - 0.492, \text{ for Nitrobacter}$$

These equations have been substantiated during recent studies by Lawrence and Brown (21) and show an approximate doubling of maximum growth rate of nitrifiers with a rise of 12°C.

McCarty (22) has proposed the following expressions for the maximum net specific growth rate ( $\mu_m$ ) for nitrifiers, as related to temperature ranges:

$$(5) \mu_m (T) = 0.5e^{0.07 (T-12)} \quad (10^\circ\text{C} < T < 30^\circ\text{C})$$

$$(6) \mu_m (T) = 0.25e^{0.18 (T-10)} \quad (5^\circ\text{C} < T < 10^\circ\text{C})$$

where  $\mu_m (T)$  = maximum net growth rate of nitrosomonas ( $\text{day}^{-1}$ )  
as a function of temperature (T)

Temperature, having such a direct effect on nitrogen bacteria, in turn affects overall efficiency of the nitrification process. Full-scale studies by Beckman et al (23) in 1972 showed the effects of seasonal temperature variations upon sewage treatment efficiency and found that a minimum of 15°C was required for 90% ammonia conversion under the process conditions used. The efficiency reduced to 75% at 10°C sewage temperature and further dropped to 50% at temperatures below 10°C. The author noted that during winter startup procedures, a definite acceleration in nitrification occurred as sewage temperatures rose above 13°C (12).

#### 5.3.4 Solids Retention Time (SRT)

To compensate for the effects of lower temperature, the bacterial mass (activated sludge) must be held within the system for a longer period, thus allowing sufficient time for bacterial growth (22).

As a function of nitrifier growth rate, the required Solids Retention Time (SRT) has been recognized during the past decade as a most important system parameter for nitrification. As shown in Table 3 the degree of ammonia oxidation increases with an increase in system SRT.

The expression for SRT is stated as being:

$$(1) \quad \theta_c = \frac{1}{\mu} \quad \text{where } \theta_c \text{ is SRT in days and } \mu \text{ is the growth rate of Nitrosomonas (22).}$$

In simpler terms, SRT is defined as follows:

$$\text{SRT (days)} = \frac{\text{wt. of aeration Suspended Solids}}{(\text{wt. of waste solids} + \text{wt. of suspended solids lost in effluent})/\text{day}}$$

SRT has also been referred to as being Sludge Age by recent authors (24,25), but must not be confused with early Sludge Age expressions proposed by Gould and/or Mohle (26).

TABLE 3  
MINIMUM SOLIDS RETENTION TIME AND DETENTION TIME  
FOR ONSET OF NITRIFICATION UNDER OPTIMAL CONDITIONS

(McCarty, 1973) (22)

T °C	* $\mu_m$ days <sup>-1</sup>	$\theta_c^m$ days	** $t_m$ hours				
			1000 mg/l MLSS	1500 mg/l MLSS	2000 mg/l MLSS	3000 mg/l MLSS	4000 mg/l MLSS
5	0.10	10	18	12	9	6	4.5
10	0.25	4.0	7	5	4	2.4	1.8
15	0.35	2.9	5	3.4	2.6	1.7	1.3
20	0.50	2.0	3.6	2.4	1.8	1.2	0.9
25	0.71	1.4	2.5	1.7	1.3	0.8	0.6
30	1.0	1.0	1.8	1.2	0.9	0.6	0.5

\* Calculated from Equations 1 and 2.

\*\* Assuming BOD<sub>5</sub> entering aeration tank is 150 mg/l and increase in suspended solids during aeration is 0.5 mg MLSS per mg BOD<sub>5</sub>, then  $t_m = 24(0.5)(150)/\mu_m X$ . Actually, the increase in MLSS is a complex function of BOD<sub>5</sub>,  $\mu_m$ ,  $\theta_c^m$ , and other factors, but the value of 0.5 allows a rough estimate to be made.



A recent paper by Sherrard and Lawrence (25) described SRT as being Mean Cell Residence Time (MCRT) and also referred to it as sludge age. Burchett and Ichobanoglous (27) have proposed the use of MCRT for the control of the activated sludge process and describe the required equipment to do so. In this case, the secondary clarifier sludge weight was added to the aeration mixed liquor portion and termed total system solids, giving the following expression:

$$(3) \quad \text{MCRT} = \frac{\text{Total System Solids}}{\text{(days) (wasted + effl. solids)/day}}$$

The limitation of this approach for design purposes, particularly for nitrification, is that the secondary clarifier usually does not offer additional oxidation of carbonaceous and nitrogenous compounds. The reverse is often the case, as a reduction of oxidized compounds (nitrite and nitrate) to nitrogen gas occurs in the clarifier sludge blanket under anoxic conditions (see Section 5.4.1). Since the secondary clarifier represents a considerable portion of the system, a large error would be introduced in the calculation of required SRT. Consequently, for purposes of design and operation the author recommends the use of expression (2).

As stated earlier, the Solids Retention Time of the aeration basin influences the degree of nitrification. In reference to expression (2), this aeration time, called  $t_m$  by Downing (13), will be related to aeration basin sludge yield and wasting rate.

To calculate the minimum values of  $t_m$ , Downing (19) has proposed the following formula:

$$(4) \quad t_m = \frac{\Delta x}{\mu_m X_i} \quad \text{where } X_i = \text{conc. of MLSS in recycle stream after dilution by raw and recycle flow}$$

$\Delta x$  = increase in aeration MLSS conc. over  $X_i$

$\mu_m$  = maximum net growth rate of Nitrosomonas

The value of minimum SRT ( $\theta_c^m$ ) can then be expressed as:

$$(5) \quad \theta_c^m = \frac{1}{\mu_m} = \frac{X_1 t_m}{\Delta X} \quad (22)$$

Calculated values of minimum SRT for the onset of nitrification have been presented by McCarty (22) and are contained in Table 3 along with computed values of  $\mu_m$  (depending on temperature) and representative values of minimum aeration detention time ( $t_m$ ) at various MLSS concentrations.

The formula  $t_m = \frac{24 (0.5) (150)}{\mu_m X}$  quoted in the footnote of McCarty's table (where  $X$  = aeration MLSS) assumes an influent  $BOD_5$  of 150 mg/l and a MLSS yield of 0.5 mg per mg  $BOD_5$ . Assuming a probable 20 mg/l ammonia-N concentration for domestic wastewater, the  $BOD_5$  to  $NH_4$ -N ratio in this case would be 150/20 or approximately 7/1. A sludge yield factor for nitrogenous oxidation has been stated as being 0.15 mg cells/mg  $NH_4$ -N oxidized (5). Comparing the sludge yields in each case, it is seen that the nitrogenous fraction ( $20 \times 0.15$ ) is insignificant when compared to the carbonaceous fraction ( $150 \times 0.5$ ).

Conversely, if a wastewater has a low  $BOD_5/NH_4$ -N ratio then the sludge yield by the nitrogenous fraction becomes significant. The expression for  $t_m$  then is:

$$t_m = \frac{24 (0.5 \times BOD_5) + (0.15 \times NH_4-N)}{\mu_m X}$$

and the nitrogenous fraction is included. The above formula would be utilized for designing or operating the second stage of a two-stage separate sludge system or for a wastewater having a low carbonaceous to nitrogenous ratio.

Since the values of SRT in Table 3 are minimum for the onset of nitrification, a safety factor must be applied to guarantee full nitrification. McCarty (22) suggests that a safety factor of between 3 and 10 times the minimum SRT (Table 3) will have to be incorporated, but wastewaters containing inhibitors to nitrification bacteria may require longer system SRT for adequate treatment. A further reference summary of data obtained by various investigators is presented in Table 4, in which effluent ammonia is equated with system temperature and SRT.

There are theoretical means by which a safety factor can be obtained, such as outlined in the EPA Nitrogen Design Manual (5), but it is the opinion of the author that a reliable safety factor can only be determined by on-site treatability studies.

#### 5.3.5 Organic and Volumetric Loading Ratio

The ratio of organic matter added to a given process biomass will influence the degree of carbonaceous and nitrogenous removal. Ratios are expressed either weight to weight as  $\text{lbs BOD}_5$  applied/day per lb aeration solids (F/M) or volumetrically as  $\text{lbs BOD}_5$ /day per 1,000 cu ft aeration volume. Using these terms, limits have been established that ensure nitrification (28). As illustrated in Figure 10, carbonaceous and nitrogenous removal is enhanced as organic loading (F/M) is reduced with nitrification appearing more significant at the lower end of the range. Sludge age is increased by decreasing F/M (Figure 10) and this further amplifies the need of long SRT for total oxidation.

TABLE 4  
SUMMARY OF RESULTS FROM NITRIFICATION STUDIES

Author	Process Type	Sewage T°C	Aeration Detention hours	Aeration MLSS mg/l	SRT days	Effl. NH <sub>3</sub> -N mg/l
Beckman et al Michigan, 1972 (23)	Single-Stage Combined Full- Scale Plant	10-18	6.3 to 7.5	2000	4 - 6	0.7 - 3.6
Wuhrmann Switzerland, 1968 (49)	Single-Stage Combined Followed by Denitrification	5-18	1.7 to 2.5	6000	4	1 - 13
Downing et al, 1961 (13)	Pilot Plant 400 gal High Ammonia Waste	16-18	4	2500 - 7000	4	1 - 60
Zenc et al Chicago, 1974 (66)	Single-Stage Combined Full- Scale Extended Aeration	7-16	11	3400	-	0.2
Sutton et al Ontario, 1975 (4)	Two-Stage, Separate Pilot Plant	5-25	1.5 & 2.6	-	4 - 10	≤ 1 - 4
Poduska and Andrews 1975 (20)	Single-Stage, Combined Lab Study	23	11	2700	10	2
Smith (author) Ontario, 1975 (10)	Single-Stage, Combined Followed by Denitrification	10-20	6 & 8	2200	11 - 40	0.1 - 0.9
Black, 1972 (9) Ontario	Single-Stage Combined Full- Scale with Lime	5-22	6.7	1500	≈ 4	0.5
Johnson & Schroeper 1964 (65)	Single-Stage Combined Lab Study	20	3.4	4760	3	5
Hall, 1974 (29)	Two-Stage Combined Lab Study	8-16	total 9.2	2350 - 4200	-	≤ 1

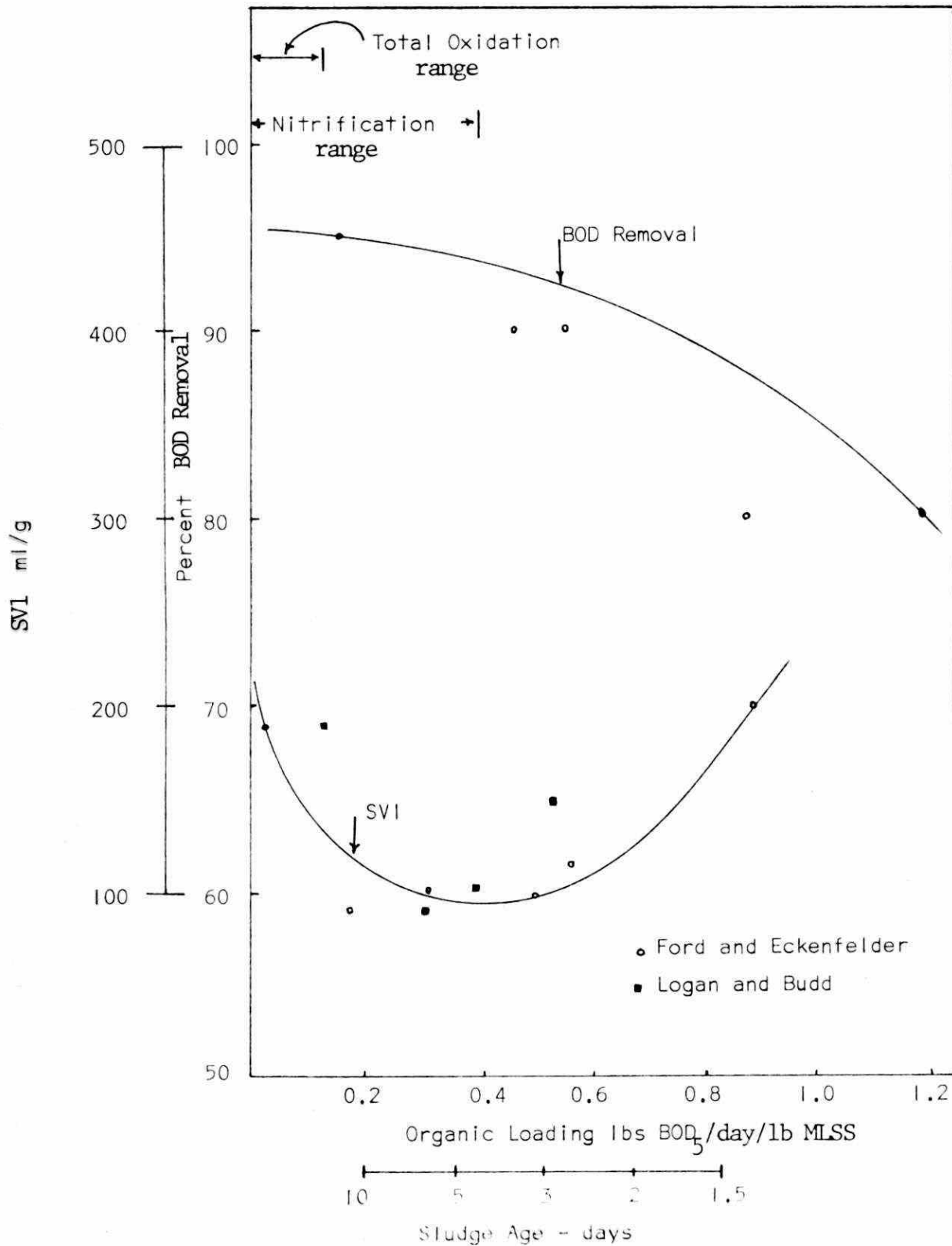
a) Aeration Detention Based on Influent Flow Only

b) Lower Values Effluent NH<sub>3</sub>-N Correspond to Higher MLSS, SRT and Detention Time

FIGURE 10

EFFECT OF ORGANIC LOADING ON PROCESS EFFICIENCY

(UNIVERSITY OF TEXAS) (28)



Beckman (23) found that an F/M range of 0.25 - 0.4 lb BOD/day/lb aer. VSS was not adequate to maintain full nitrification using a single sludge system. A wastewater temperature range of 7 to 18°C was studied and ammonia removals were 50 and 95% respectively with sludge ages of between 4 and 12 days. Better results were achieved in experiments by Hall (29) with a two-stage combined sludge system. In these studies, consistent low F/M's of between 0.11 and 0.17 (based on aer. VSS) produced effluent ammonia-N concentrations of 1 mg/l or less at a range of 7 to 16°C. MOE study processes (12) were designed to operate between 0.15 and 0.25 lbs BOD<sub>5</sub>/lb aer. VSS and a high degree of nitrification was accomplished as long as other parameters such as minimum aeration DO were maintained.

Two-stage, separate sludge systems such as evaluated by Sutton and Jank (4), normally have a high rate (high F/M) first stage followed by a low rate (low F/M) second stage. An example of this type of arrangement is the recent expansion of the Cleveland Southerly plant (30) based on a previous (unpublished) pilot plant study. The design plan calls for a first stage F/M of 1.1 and a second stage of 0.12.

Volumetric loading ranges for high rate, conventional and extended aeration are of the order 50, 20-50 and 20 lb BOD<sub>5</sub> per 1,000 cu ft aeration volume, respectively. The lower range of extended systems, giving low F/M and high SRT's (15 days), will normally result in a high degree of nitrification under a wide range of wastewater temperatures (31).

The disadvantages in using organic and volumetric loadings for operation of nitrification systems is that the BOD<sub>5</sub> test required for their calculation takes five days, thus making daily control of these process parameters virtually impossible.

From a design point of view, the process loading to achieve a required effluent quality will depend on the type and expected temperature range of the wastewater to be treated. Consequently, a treatability study similar to that for SRT would be required at each site.

#### 5.4 Secondary Clarification and Sludge Recycle

##### 5.4.1 Single-Stage Process

The secondary clarifier can help maintain system SRT if adequately sized. Moreover, if low levels of BOD are required in plant effluents, stringent requirements must be imposed upon suspended solids removal. In some areas of Ontario, tertiary filtration is being applied to polish effluents to achieve receiving water requirements and their effectiveness depends greatly on the quality of the secondary clarifier overflow.

Effluent quality will also depend on aeration Sludge Volume Index (SVI) as sedimentation is deterred when values rise over 200 ml/g and MLSS may be lost over the clarifier weir during high flow periods. Frequent loss of solids will also lower the system SRT and decrease the nitrification efficiency.

The addition of harsh chemicals such as sodium hypochlorite to treat a high SVI (poor settling) sludge endangers the nitrifying bacteria population (12,32), thus their use would have to be limited. This being the case, the design and size of a secondary clarifier in a single-stage system should be adequate to handle a low density sludge if it develops. Filamentous organisms commonly associated with a high SVI or bulky sludge do not normally affect nitrification efficiency adversely, in fact, some varieties such as *Escherichia coli* (E-coli) have been found to be present in large numbers only when nitrification occurs (33).

To obtain the benefits of longer process SRT's at low wastewater temperatures, solids concentration by decreased sludge wasting will be required in the aeration basin. This in turn will impose an increased solids loading on the clarifier stage and probably result in a deteriorated effluent unless the clarifier is capable of handling the extra solids loading.

Extensive full-scale studies by Pflanz (34) in Germany over a period of two years revealed how factors such as SVI, solids loading, temperature, overflow rate and clarifier design affect the secondary clarifier effluent quality. These experiments studying the effects of raised SVI on rectangular clarifier performance were conducted by withdrawing the addition of lime and iron sulfate from an activated sludge process. A rise in effluent solids resulted from increased SVI's (see Table 5 ).

TABLE 5  
VARIATIONS IN CLARIFIER EFFLUENT SOLIDS CONCENTRATION  
WITH CHANGES IN SVI  
(Pflanz, 1969)<sup>(34)</sup>

<u>Aeration MLSS (g/l)</u>	<u>Sludge SVI (ml/g)</u>	<u>Effl. SS mg/l</u>
2.2	141	13
2.1	171	20
1.8	255	39



As illustrated in Figure 11, a relationship was seen between solids loading, overflow rate and resultant clarifier overflow suspended solids at various sewage temperatures. Pflanz's results agreed with earlier experiments performed by Keefer in 1962 (35), where an increase in secondary effluent suspended solids was noted at each increased solids loading range with a corresponding drop in temperature. Out of these experiments, the importance of solids loading on clarifier performance was realized particularly in regard to levels of SVI. A summary of recommended clarifier solids loading rate at various SVI's to achieve a level of 30 mg/l SS effluent quality is presented in Table 6.

TABLE 6  
EFFECT OF SVI UPON ALLOWABLE CLARIFIER SOLIDS LOADING  
(Pflanz 1969) (34)

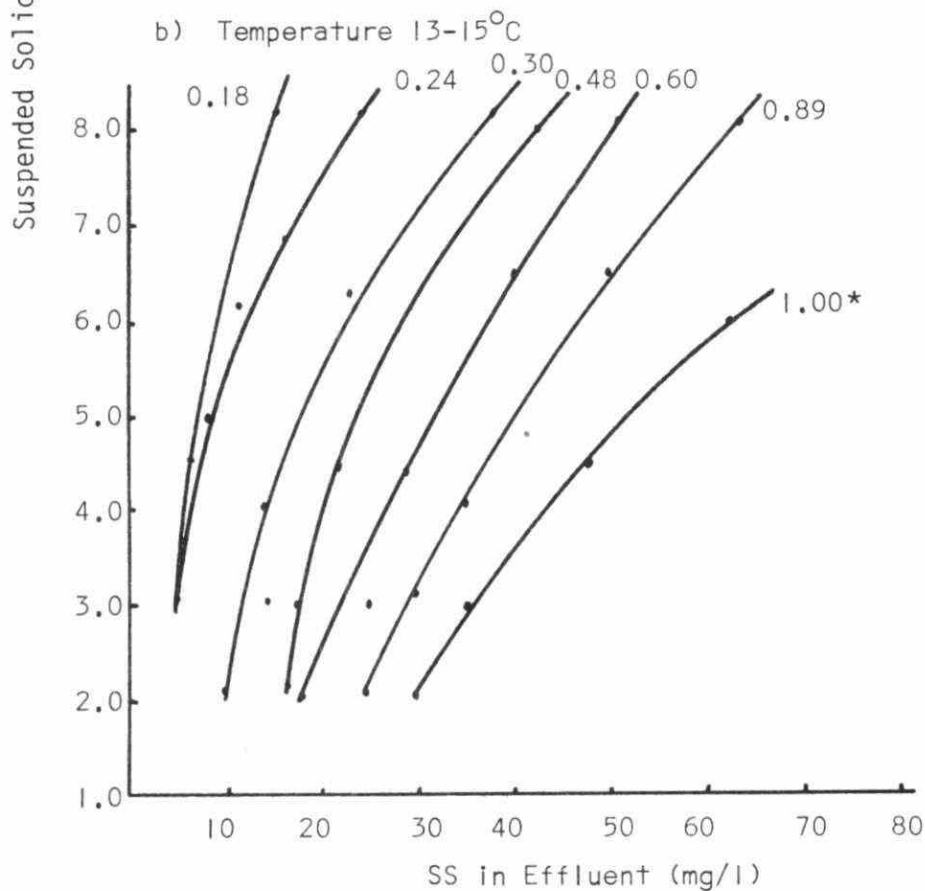
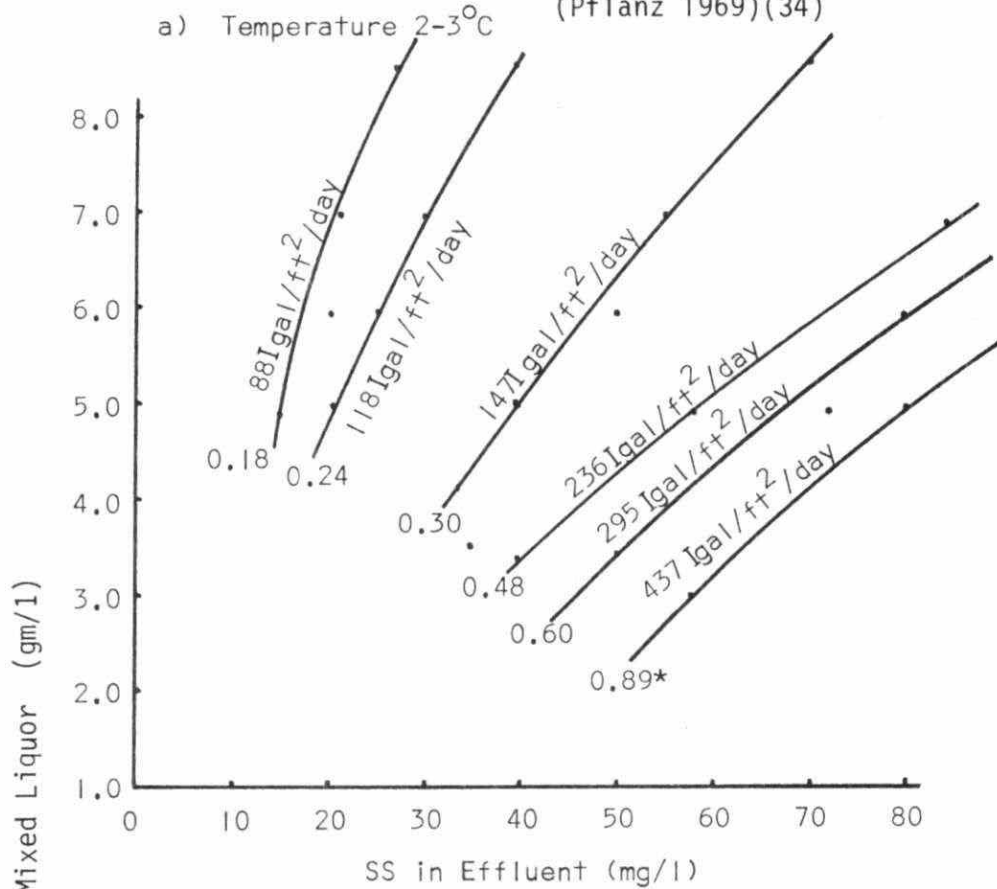
<u>SVI (ml/g)</u>	
100	3.5 kg SS/m <sup>2</sup> /hr (0.7 lbs SS/ft <sup>2</sup> /hr)
200	1.1 to 1.3 Kg SS/m <sup>2</sup> /hr (0.2 to 0.3 lbs SS/ft <sup>2</sup> /hr)
300	0.8 to 1.1 Kg SS/m <sup>2</sup> /hr (0.15 to 0.2 lbs SS/ft <sup>2</sup> /hr)

Similar experiments at the Brampton OEF (12) on secondary clarification have shown that with a SVI of 460 ml/g a maximum solids loading of 0.3 lbs SS/ft<sup>2</sup>/hr (1.3 Kg SS/m<sup>2</sup>/hr) could not be exceeded without a deterioration in effluent quality. The same process having a SVI of 160 ml/g allowed a maximum solids loading value of 0.7 lb SS/ft<sup>2</sup>/hr (3.5 Kg SS/m<sup>2</sup>/hr). The results do not entirely agree with those

NOTE: Clarifier solids loading values do not include return sludge flow.

FIGURE 11

CLARIFIER OVERFLOW RATE VS SUSPENDED SOLIDS INFLUENT AND EFFLUENT  
(Pfanz 1969)(34)



\* Values expressed in m/hr.

obtained by Pflanz but similarities can be noted as to the effects of SVI on maximum clarifier solids loading. Moreover, the experiments at the OEF were performed on a circular clarifier 60 ft in diameter by 10 ft deep and shallow clarifiers such as these often suffer greater sludge blanket disturbances due to density currents.

The United States Environmental Protection Agency (EPA) (36) recommends a permissible secondary clarifier overflow rate of 400-600 US g/ft<sup>2</sup>/day (330-500 Ig/ft<sup>2</sup>/d) (16-25 m<sup>3</sup>/m<sup>2</sup>/day) up to a maximum clarifier influent SS of 2500 mg/l for nitrification plants. To facilitate less sludge blanket interference, clarifiers having more than 10 ft (3 m) (36) sidewall depth are recommended. The EPA states 12 feet (3.6 m) (36) as the minimum depth.

Investigations by Schmidt-Bregas (37) have shown that water depth to length ratios should be 1:20 to 1:35 for rectangular clarifiers. Circular clarifier depth to radius ratios of 1:6 to 1:8 were also recommended.

The Newmarket WPCP (12) operated well as a nitrification plant with mean diurnal secondary clarifier overflow rates of 816 IG/ft<sup>2</sup>/day (40 m<sup>3</sup>/m<sup>2</sup>/day) with a solids loading of 0.7 lbs/ft<sup>2</sup>/hr (3.4 Kg/m<sup>2</sup>/hr) and produced an effluent suspended solids of 22 mg/l when SVI's were less than 150 ml/g, but bulking occurred when SVI's were above 200.

Boyko and Rupke (38) determined that a maximum overflow rate of 700 IG/ft<sup>2</sup>/day (34 m<sup>3</sup>/m<sup>2</sup>/day) was required in a conventional activated sludge system to maintain a secondary clarifier effluent quality of less than 15 mg/l SS when using phosphorus removal chemicals as part of a post-precipitation process. Consequently, chemical addition in this manner to a nitrification process can also dictate the maximum overflow rate.

The recirculation rate of settled clarifier sludge to the aeration basin is of the utmost importance to the maintenance of nitrification. At high rates of recirculation (more than 80% of raw sewage flow), a more consistent concentration of suspended solids can be maintained in the aeration basin and thus stable conditions are better sustained (18). The rapid withdrawal of settled sludge ensures that this sludge does not rest in the clarifier and become anoxic. Under anoxic conditions, the nitrate compounds developed by the aeration basin reactions will reduce to nitrogen gas (denitrification) which will float floc particles to the clarifier surface. It has been recommended (5) that surface skimmers be used to remove denitrification scum which may be produced with reduced clarifier DO during warm weather. Operating at a minimum of 80% recycle flow based on raw sewage input should keep scum formation to a minimum under Ontario's climatic conditions.

The exception to this will be in extended aeration systems in which skimming is normally provided in the secondary clarifier to remove non-biodegradable material which accumulates in this type of process.

In view of the foregoing data, the required secondary clarifier solids loading and overflow rates are  $<0.5 \text{ lbs/ft}^2/\text{hr}$  (or  $24 \text{ lbs/ft}^2/\text{day}$  ( $117 \text{ Kg/m}^2/\text{d}$ ) including 100% recycle) and  $<600 \text{ Ig/ft}^2/\text{day}$  ( $<29 \text{ m}^3/\text{m}^2/\text{d}$ ).

#### 5.4.2 Two-Stage Processes

In a two-stage, separate sludge mode of nitrification process, individual consideration must be given to each stage. In a combined sludge arrangement with plug flow, carbonaceous oxidation is accomplished in the first half of the aeration basin and the biochemical reactions gradually shift to the nitrogenous phase until at the basin end full nitrification is accomplished. The inter-relationship of carbon removal

and the nitrogen conversion is such that the heterotrophic bacteria in utilizing organic carbon provide, through their metabolism, an inorganic carbon source ( $\text{CO}_2$ ) for the nitrifying autotrophic bacteria (39). The two-stage system is primarily designed to split the carbonaceous and nitrogenous removal functions into two separate units, but if the first stage does not have sufficient organic loading, nitrification may also take place. If this happens to any degree, the second-stage nitrification reactions could suffer due to low levels of ammonia and carbon.

To help prevent any appreciable nitrification in the first stage, this unit must be designed as a high rate process preferably with complete mixing and high clarifier overflow rates (18). Most of the sludge production occurs in the first stage and most of the wasting is provided from there. A much slower biomass growth rate results in the second stage aeration basin with nitrification reactions. During periods of low organic loading to the first stage, a resultant drop in nitrification efficiency may occur in the second stage due to process starvation. Quantities of primary effluent and/or biological sludge from the first stage may then have to be bypassed to the second stage aeration basin, therefore, facilities for this option should be provided.

Studies by Sutton et al (4) on a two-stage constant flow process involved the use of two clarifiers, both having an overflow of  $823 \text{ I gal/ft}^2/\text{day}$  ( $40 \text{ m}^3/\text{m}^2/\text{d}$ ). The solids loadings to the clarifiers were  $1 \text{ lb SS/ft}^2/\text{hr}$  ( $4.8 \text{ Kg SS/m}^2/\text{hr}$ ) and  $1.3 \text{ lb SS/ft}^2/\text{hr}$  ( $6.3 \text{ Kg SS/m}^2/\text{hr}$ ) for the first and second stage respectively. The suspended solids concentration from the first clarifier averaged  $30 \text{ mg/l}$  and under these conditions the second stage aeration basin lacked in sludge growth. Reseeding of the

second stage aeration with first stage sludge was necessary to maintain second stage SRT of 7 days. A sludge recycle rate of 125% of influent flow was maintained in both stages (personal communication).

Observations by Mulbarger (40) experimenting with the three stage system for nitrification and denitrification showed that the addition of sodium aluminate to the first stage aeration at a dosage of 14 mg/l (as  $Al^{+3}$ ) maintained an SVI of 40 ml/g. Under these conditions, both clarifier overflow rates in the two-stage nitrification plant approximated 1000 I gal/ft<sup>2</sup>/day (48 m<sup>3</sup>/m<sup>2</sup>/day) and produced clarifier SS concentrations of 40 and 18 mg/l for first and second stages respectively. An effluent ammonia concentration of 2.2 mg/l was produced from the nitrification stage.

The USEPA design manual (18) recommends that the first stage clarifier have a minimum 2.4 hours detention with a 1000 I gal/ft<sup>2</sup>/day (48 m<sup>3</sup>/m<sup>2</sup>/day) overflow rate and a second stage clarifier detention of 3.0 hours with a 660 I gal/ft<sup>2</sup>/day (32 m<sup>3</sup>/m<sup>2</sup>/day) overflow rate. Also, a maximum BOD<sub>5</sub> of 40 to 50 mg/l is suggested in the first stage clarifier effluent (36).

#### 5.4.3 Summary

As seen in Section 5, there are a number of factors which affect the efficiency of the nitrification process. The relative importance of each is still to be clearly established under full-scale conditions and is being studied by investigators in the wastewater treatment field. Consequently, the criteria based on some of the factors, such as the role of toxic substances under full-scale conditions may be subject to revision. A summary of the current thinking is given in Table 7.

TABLE 7

PARAMETERS AFFECTING THE NITRIFICATION PROCESS  
AND SUGGESTED DESIGN RECOMMENDATIONS

1. Waste and Flow Variation - Flow equalization if peaking factor frequently exceeds 1.5.
2. Aeration pH and Alkalinity - 6.5 to 8.5 - optimum 8.3  
minimum effluent of 50 mg/l as  $\text{CaCO}_3$
3. Aeration  $\text{O}_2$  requirements (based on average daily  $\text{BOD}_5$  and TKN). Peaking factor applied to  $\text{NH}_4\text{-N}$  for conventional.
  - Nitrogenous - 4.6 lbs  $\text{O}_2$ /lb  $\text{NH}_4\text{-N}$  -Conventional
  - 4.6 lbs  $\text{O}_2$ /lb TKN -Extended
  - Carbonaceous- 0.8 lb  $\text{O}_2$ /lb  $\text{BOD}_5$  -High Rate
  - 1.0 lb  $\text{O}_2$ /lb  $\text{BOD}_5$  -Conventional
  - 1.5 lb  $\text{O}_2$ /lb  $\text{BOD}_5$  -Extended
  - Minimum DO of 2 mg/l in all areas of the aeration tank.
4. Solids Retention Time (SRT) - For 1 mg/l effluent ammonia nitrogen
  - > 2 days at  $20^\circ\text{C}$
  - >10 days at  $5^\circ\text{C}$
  - Safety factor to be determined via treatability studies.
5. F/M Ratio - 0.05 to 0.25 lb  $\text{BOD}_5$ /lb MLVSS
6. Secondary Clarifier
  - (a) High Rate (First Stage)
    - Detention (hrs) - 2.5
    - Overflow rate - 1000  $\text{lg}/\text{ft}^2/\text{day}$  (PDWF)\*  
(49  $\text{m}^3/\text{m}^2/\text{day}$ )
  - (b) Second Stage or Combined System
    - Detention (hrs) - >3.0 hrs
    - Overflow rate - <600  $\text{lg}/\text{ft}^2/\text{day}$  (PDWF)  
(29  $\text{m}^3/\text{m}^2/\text{day}$ )
    - Solids Loading - <25  $\text{lbs}/\text{ft}^2/\text{day}$  (PDWF)  
(122  $\text{Kg}/\text{m}^2/\text{day}$ )  
(including 100% recycle flow).
    - Minimum sidewall depth of 12 ft (3.6 m)
7. Sludge Recycle Rate - Minimum of 80% of influent flow (ADWF)\*\*

\* The average peak 4 hour flow during dry weather.

\*\* The average daily flow expected during dry weather.

## 6. PROCESS MONITORING FACILITIES AND TECHNIQUES

Biological systems respond slowly, particularly with reference to nitrification bacteria and their relatively slow growth rates. If an activated sludge process is severely upset due to the addition of a toxic wastewater, equipment failure or improper process operation, the recovery time for re-establishment of nitrification is usually much longer than that for carbonaceous oxidation (12). Moreover, recovery will be further delayed under low wastewater temperature conditions and slow bacterial reproduction rate. Consequently, constant facility monitoring, analyses and process control are necessary to maintain consistent nitrification under all conditions.

### 6.1 Influent Monitoring Devices

Continuous monitoring of influent parameters is imperative when nitrification is required on industrial applications or combined domestic-industrial wastewater with varying characteristics. Recording pH or continuous respirometers are useful in this respect and can be used to trigger an alarm system when unwanted substances are present in the process influent. A viable system to indicate potential toxicity would be to have a continuous on-stream respirometer which would trigger an influent bypass stream to a holding tank when the oxygen utilization rate of the influent wastewater falls below an arbitrary value. A recording pH meter and alarm system can be utilized instead of the aforementioned instrument if toxic wastes are absent and only a buffering system needs to be activated, i.e. aeration pH control by sodium hydroxide addition.



When using chemicals for phosphorus removal prior to biological treatment, a pH electrode feed-control and/or alarm unit could be provided to prevent overdosing and consequent carryover of excess chemical to the nitrification process.

A continuous readout of aeration basin dissolved oxygen is recommended. The probe must be located to indicate a mean chamber oxygen value. This can be accomplished by using a large rugged DO probe, either of submersible or floating type, which relays a signal to a monitor placed in the plant laboratory or office. A low DO alarm can also be included with this system. In addition, this equipment could also be part of an automated aeration control system to control the level of mixed liquor DO. This method is discussed in Appendix 12.3.

## 6.2 Sampling and Wasting

In reference to a conventional facility, twenty-four hour composite sampling should be carried out on raw sewage, primary and secondary clarifier effluents. Flow proportional samplers, which vary the sample aliquot taken at each interval depending on flow, may be required if a stringent loading survey of facility or receiving stream is sought.

Daily grab and/or composite samples of aeration and waste activated sludge should be taken depending on variability of MLSS at a given plant. The variability of MLSS will be influenced by the diurnal fluctuation of influent wastewater concentration and flow. For a plug-flow aeration basin, Burchett and Ichobanoglous (27) have advocated

the use of a manifold sampler which derives a composite via separate bay sampling. A device such as this would be implemented for accurate control of SRT if wasting of aeration mixed liquor is practiced.

To facilitate better regulation of solids, continuous wasting is mandatory whether from the commonly used return sludge line or from the aeration basin. A lack of adequate wasting facility is a common design fault in activated sludge systems. This can be corrected easily by installing a variable speed sludge pump with a metering system to a return sludge line or aeration basin.

### 6.3 Analyses and Tests

On-site chemical analyses should be complete enough so that the operator can make the necessary process adjustments to achieve maximum nitrification.  $BOD_5$  and/or COD analyses on composite aeration basin influent, plus aeration solids determination can facilitate the maintenance of proper F/M ratios, SRT and resultant sludge wasting schedule. Since there is considerable lag in obtaining  $BOD_5$  results, two alternate procedures can be considered. The first is the use of an automatic  $BOD_5$  apparatus which extrapolates results within 24 hours, the second is to determine the relationship of  $BOD_5$  and COD in the plant influent and effluent and use the COD value to control the process.

Nitrification efficiency can be monitored by performing ammonia-nitrogen analyses on the influent and effluent composite samples 3 days per week. Nitrate and nitrite should be analyzed on-site one or two times to ensure nitrogenous oxidation is proceeding fully to nitrate. Levels of nitrite above 2 mg/l usually indicate a disruption in the nitrification reactions.

On-site analyses of suspended solids is necessary, particularly if levels of suspended solids below 10 mg/l are required in the final effluent. At these low levels, a minimum of 1 litre sample is required to obtain an accurate suspended solids result. The removal efficiency of insoluble organic nitrogen as floc by a tertiary filtration process can be approximated from the suspended solids or turbidity analyses if a correlation with organic nitrogen (Total Kjeldahl minus ammonia) is determined.

To control the SRT, aeration and waste mixed liquor suspended solids should be determined daily in order to regulate the sludge wasting schedule.

To monitor the biological aspect of the nitrification process, an oxygen utilization apparatus (respiration rate) and a microscope are invaluable. The oxygen utilization rate will indicate the activity of the bacteria and the organic loading. Usually an  $O_2$  utilization rate ( $\text{mg } O_2/\text{l/hr}$ ) will match the volumetric process loading in  $\text{lbs BOD}_5$  added per 1000 cu ft/day for a domestic waste. When operating a two-stage plant, this test can facilitate the balancing of organic input to each stage.

Microscopic examination of process protozoa if performed on a regular basis can often indicate when toxic conditions arise in the process. Used as indicators, these organisms, depending on type and distribution, can also indicate the stability or state of evolution of an aeration ML. It may then be possible to relate these observations over a period of time to nitrification performance at a given site. Techniques for using protozoa as process indicators are outlined in the MOE Activated Sludge Course Manual (41).

Maintaining a minimum aeration DO of 2 mg/l (41) will necessitate a DO survey to be part of the daily routine of an operator if automated readout is not available. This survey should be conducted at least during the peak loading period when ammonia is more apt to escape the process due to hydraulics and high oxygen demand. The Winkler procedure for DO analysis is acceptable but a DO meter, if calibrated frequently, is just as accurate and can be used for oxygen uptake tests (41).

There are a number of kits commercially available which are capable of analyzing a wide variety of substances including nitrogen and phosphorus. The accuracy of these kits are subject to criticism for certain analyses, but the advantage of fresh sample, on-site analyses may outweigh this factor. Current investigations by the MOE Central Laboratory on sample stability and preservation (42) have shown that concentrations and forms of carbonaceous and nitrogenous substances can change dramatically upon storage.

WPCP Laboratory Facilities

To facilitate adequate process control, the following lab equipment should be available to the plant staff for on-site analyses:

	<u>Approximate Cost*</u>
1. Drying oven and muffle furnace for suspended and volatile solids analyses	\$ 436
2. Field type dissolved oxygen probe and meter	108
3. Incubator and associated apparatus for doing BOD analyses	2,976
4. Kit and chemicals for ammonia analyses	573
5. Kit for nitrite, nitrate and soluble phosphorus analyses	950
6. Microscope having the capability of up to X400 magnification with an electrical light source	1,482
7. Glassware associated with the above tests including 1 litre graduate cylinders for solids settling measurements	400
8. pH meter and accessories	315
9. Balance	800
	<hr/>
	\$8,040

Options

a) Apparatus and chemicals for doing COD analyses

or

b) Apparatus for doing automatic BOD analyses

The analyses using equipment items (5) and (6) above could also be performed at a nearby laboratory if the samples are prepared for shipment. The stabilization of samples for ammonia, nitrite and nitrate and soluble phosphorus can be greatly improved by filtering them through two layers of glass-fibre filter paper and storing in a refrigerator (12).

\* Equipment for a laboratory servicing a 1 MGD WPCP.

#### 6.4 TOD Monitoring

As expressed in the introduction to this report, the WPCP effluent discharge criteria may be stated in terms of allowable TOD. For example, a facility may be required to meet a certain maximum pounds TOD discharged per day such as proposed in the Thames River report (1). To the author's knowledge, there are no TOD analyzers developed as yet that can accurately measure values below 50 mg/l, which would be the range in samples from secondary or tertiary effluents. The TOD value must then be calculated from a formula such as follows:

$$1. \text{ TOD} = \text{NOD} + \text{CAR BOD}$$

$$= 4.57 \times \text{TKN} + R \times \text{BOD}_5$$

$$\text{where: } R = \frac{\text{BOD}_{20}}{\text{BOD}_5} \quad (\text{Carbonaceous BOD only})$$

For example, a plant's secondary effluent normally contains 4 mg/l TKN, and 15 mg/l  $\text{BOD}_5$  at a plant flow of 10 MIGPD. Effluent analysis of 20 day BOD ( $\text{BOD}_{20}$ ) has shown a value of 45 mg/l, therefore, an R value of 3 ( $45 \div 15$ ) is given. Using this information, the pounds TOD discharged per day would be calculated as follows:

$$\begin{aligned} \text{lbs TOD/day} &= (4.57 \times 4 + 3 \times 15) (10 \times 10) \\ &= 6,328 \end{aligned}$$

NOTE: Only carbonaceous oxidation is required in the  $\text{BOD}_{20}$  test. Therefore, nitrification must be accounted for or suppressed by chemical addition.

## 7. NITRIFICATION AND PHOSPHORUS REMOVAL

It is quite probable that phosphorus removal by the addition of chemicals will be required at the facilities being evaluated for nitrification. Consequently, the effects of these chemicals on nitrification reactions and on sludge yield and composition should be considered.

### 7.1 Toxicity

A thesis prepared by Kitay (43) on the effects of addition of calcium oxide ( $\text{CaO}$ ), aluminum sulphate ( $\text{Al}_2(\text{SO}_4)_3$ ) and ferric chloride ( $\text{FeCl}_3$ ) for P-removal at concentrations above 200 mg/l showed that the metal components of these compounds were not toxic to nitrification bacteria. The resultant low ( $< 6.5$ ) or high pH ( $> 9.0$ ) of the mixed liquor with the addition of these compounds was, however, found to be the inhibiting factor for nitrification reactions.

Phosphorus removal studies by the author (44) in which 200 mg/l of  $\text{Ca}(\text{OH})_2$  with 5 mg/l (as  $\text{Fe}^{+2}$ ) ferrous sulphate were added to the raw sewage at Newmarket showed no effects on process nitrification. Continuing experiments on the nitrification-denitrification process at the OEF (12), which involved the addition of ferric chloride to the activated sludge over a 10 week period at dosages ranging from 7 to 30 mg/l as  $\text{Fe}^{+3}$ , produced no effect on nitrification.

Conversely, investigations by Sutton et al (4) showed some deterioration of process nitrification when the ferric chloride dosage to the aeration basin was raised from 15 to 30 mg/l as  $\text{Fe}^{+3}$ .

Two distinct differences between these studies were that the wastewater alkalinity and basin MLSS were higher in the OEF process and probably offered more buffering to the ferric chloride acidity.

P-removal chemicals often contain heavy metals and could indirectly offer toxicity to nitrification bacteria. The concentrations of these metals such as zinc, copper and chromium can vary between batches and will depend on the source of raw materials and manufacturing techniques. MOE chemical analyses of various P-removal chemicals listed in Table 8 show particularly high levels of zinc in iron salts, although these levels are considerably lower in the specially prepared batch of supplier B\*.

Generally, the addition of these chemicals will not directly impose toxicity to the nitrification system due to sewage dilution, but accumulation of heavy metals within floc particles and digester sludge could eventually pose problems. (See Appendix 12.2.1)

## 7.2 Effect on Alkalinity

Studies by Stepko and Shannon (45) showed a loss in wastewater alkalinity of 5.3 to 13.5 lbs as  $\text{CaCO}_3$  per lb  $\text{Fe}^{+3}$  and 6.0 to 9.0 lbs as  $\text{CaCO}_3$  per lb  $\text{Al}^{+3}$  added, respectively.

Since there was no allowance for alkalinity lost through nitrification and quantity of unused phosphorus removal chemical, these values probably are too high.

To accurately assess the alkalinity loss due to metal ion addition, the following expression may be used:

$$A_{Lp} = \frac{(A_i - A_e) - A_{Ln}}{M_i - M_e} \quad (1)$$

where:  $A_{Lp}$  = is the alkalinity loss mg/l (as  $\text{CaCO}_3$ ) across the system due to metal addition

$A_i$  = influent alkalinity

$A_e$  = effluent alkalinity

$A_{Ln}$  = alkalinity loss due to nitrification

$M_i$  = influent soluble metal ion conc., mg/l

$M_e$  = effluent soluble metal ion conc.



TABLE 8

TOTAL HEAVY METAL ANALYSES OF PHOSPHORUS REMOVAL CHEMICALS  
(mg/l)

Chemical	Supplier	As	Cd	Co	Cr	Cu	Hg( $\mu$ g/l)	Mo	Ni	Pb	Se	Zn
FeCl <sub>3</sub>	A	<150	<80	<4	80	15	ND	ND	40	275	ND	400
	B	8.4	20.2	19	180	160	<.01	40	50	2100	<2	14,500
	*B		0.27		22	20	0.011		12	55		410
	C		1.4	31.2	661	11.8	<.001	17	41.8	113	0.042	260
Alum	D	<2	0	<1	<1	1.0	0.004	<1	<1	0.05	<1	<1
	E	0.01	0.68	0.37	47	0.75	<0.001	0.05	1.2	0.83	<0.01	2.2
	F		0.12		0.2	0.22			0.08	1.9		1.4
FeSO <sub>4</sub>	G	<0.002	<0.1	1.4	3.3	0.5	0.006	ND	6.5	<0.1	0.12	5.6
	H				0.003							
	I	ND	0.2	10	51	240	0.17	260	29	4.6	3.2	860
FeCl <sub>2</sub>	J	<0.026	<0.1	1.6	36	5.6	0.010	ND	6.9	<0.1	0.3	1.7
	K	<10	<1	10	20	9	<10	ND	18.6	1.5	<10	2.3
Lime	L	1	0.2	0.2	0.6	3	<.02	2	3	1	ND	25

\*B - Special low heavy metal preparation.

$A_{Ln}$  can be determined using the theoretical 7.14 mg Alk lost/mg  $NH_4-N$  oxidized (5) or preferably on a given wastewater by a process evaluation without phosphorus removal.

Unpublished data from recent studies by Sutton et al (46) have shown that on either single or separate sludge systems, the alkalinity loss due to the addition of  $Fe^{+3}$  or  $Al^{+3}$  is close to the stoichiometric value. By comparing parallel operated systems, with and without phosphorus removal, a value of 2.9 lbs alkalinity loss per lb  $Fe^{+3}$  utilized resulted after allowing for nitrification and unused chemical.

Analyses of data from the Gravenhurst, Ontario WPCP by the author have indicated an alkalinity loss factor of 1.7 using ferric chloride. Current sampling of a full-scale nitrification system with and without ferric chloride addition at the OEF shows values of between 2 and 3 (47).

In areas with wastewaters of low alkalinity, (<150 mg/l as  $CaCO_3$ ) a supplement of caustic soda (NaOH) or lime (CaO) will be required to maintain aeration pH above 6.5. The cost of caustic soda often prohibits its use depending on locality, consequently lime is normally chosen. Theoretically, 0.56 lbs of CaO will produce 1 lb of  $CaCO_3$  alkalinity, therefore, phosphorus removal via primary stage lime addition preceding nitrification may appear attractive. However, experiences with this type of process at Newmarket (12, 48) indicate that lime removes trace quantities of heavy metals and this condition can result in the growth of filamentous bacteria. This results in raised SVI which in turn causes a loss in solids via secondary clarifier bulking. The problem was alleviated on adding ferrous sulphate with a reduced lime dosage to the primary stage (44). These findings agree with early studies by Wuhrmann (49) inasmuch as a lime/ferric combination gave better overall results than individual chemicals.

The use of sodium aluminate in nitrification plants has been suggested by Sutton et al (46) because alkalinity is produced with phosphorus precipitation reactions. The major drawback of this chemical is its high cost and only limited information is available regarding its performance.

To maintain a minimum recommended aeration pH of 6.5, a minimum residual alkalinity of 50 mg/l as  $\text{CaCO}_3$  is required (5).

A typical phosphorus removal/nitrification system alkalinity calculation would be as follows:

e.g. assuming influent ammonia - N of 20 mg/l and a chemical addition of 15 mg/l ferric chloride as  $\text{Fe}^{+3}$ , the total alkalinity loss using stoichiometric values would be:

$$(20 \times 7.1) + (15 \times 2) = 142 + 30 = 173 \text{ mg/l}$$

therefore, to maintain an alkalinity residual of 50 mg/l, a minimum influent level of 223 mg/l would then be required.

As stated in Section 5.3, the alkalinity lost due to nitrification can vary from 5.5 to 9.0 lbs/lb ammonia-N oxidized. This variation, dependent on biological activity, will often contribute significantly in the choice of the appropriate phosphorus removal chemical. Consequently, a process evaluation is suggested on an existing facility to determine the actual alkalinity loss particularly on a sewage of low to medium (<250) alkalinity.

8. APPROACHES TO PROCESS EVALUATION AND  
TREATABILITY STUDIES FOR DESIGN PURPOSES

It is recommended that prior to the conversion of an existing wastewater treatment plant, or before designing a new plant to include the nitrification process, treatability studies with the wastewater be carried out.

Because of the variation in the composition of wastewaters, these individual small scale tests will help to:

- (1) Identify the need for any pretreatment before nitrification or for any post-treatment;
- (2) Identify the nature of the wastewater indicating any inhibiting or toxic materials to the nitrifying bacteria. This will affect the design of equipment sizes, flow-rates, etc.;
- (3) Specify if any particular sections of an existing process require changes;
- (4) Provide, in general, the optimum facility for the nitrification in consideration of the variables discussed.

A treatability study for nitrification can be conducted by using a laboratory, pilot plant or semi-full-scale process unit. For purposes of classification, each of these units can be defined by capacity as follows:

- (a) Laboratory Unit - usually less than 100 gallons ( $0.45 \text{ m}^3$ )/day.
- (b) Pilot Plant - up to 10,000 gallons ( $45 \text{ m}^3$ )/day.
- (c) Semi-full-scale - more than 10,000 gal up to 1 mgd ( $4,500 \text{ m}^3$ /day).

Reference can be made to Appendix 12.4 for further details on the test units.

The following sections describe recommended procedures for carrying out treatability studies when designing for nitrification at a new site or at an existing facility.

#### 8.1 New Facilities

Designing a nitrification facility from raw waste information only is a difficult task because of the lack of previous treatment information. A complete survey of upstream conditions, such as sewer configuration, industrial waste input, expected temperature range and probability of stormwater influence is imperative. After this information has been gathered, the following sequence of procedures is recommended:

- (a) Twenty-four hour sequential sampling with flow monitoring and analyses of wastewater components that affect nitrification.
- (b) Full range of treatability studies involving pilot-scale primary sedimentation and activated sludge treatment.

The purpose of procedure (a) would be to determine the expected carbonaceous and nitrogenous (particularly diurnal peak) loadings. The degree of variability of flow and wastewater strength can help in deciding whether flow equalization will be needed.

The objectives of procedure (b) are as follows:

- 1. To study primary sedimentation;
- 2. To arrive at a suitable SRT to maintain a target effluent ammonia concentration.

Primary sedimentation will constitute a reduction in total oxygen demand material, therefore, primary clarification should be included as part of (b). Sequential sampling and analyses of primary effluent will then produce secondary loading limits. As an alternative method, a theoretical  $BOD_5$  removal via primary clarification may be

assumed, depending on intended detention time and overflow rates (Section 5.2). This approach may be somewhat risky, unless predetermined removals are supported by on-site batch sedimentation tests such as outlined by Oliver (50, 51).

In running a treatability study, the following procedure may be adopted:

For example, consider a receiving stream regulation that requires a domestic WPCP effluent not to exceed 1 mg/l ammonia-N from April to October. Expected wastewater temperatures during this period range from 12 to 25°C. Table 3 (Section 5.3.3) gives minimum SRT required at these temperatures of 3.5 down to 1.4 days. To facilitate rapid nitrifier growth, the best approach would be to start the treatability unit at the highest temperature and operate at an SRT possibly 3 to 5 times the minimum (4.2 to 7 days).

Following the onset of nitrification reactions, an SRT can be found at 25°C that will give constant effluent ammonia-N concentration of  $\leq 1$  mg/l. The wastewater temperature can then be lowered gradually until the recommended ammonia level is exceeded. The system SRT is then raised by reducing wasting and an optimum can be sought at the new temperature. By following this pattern, an optimum SRT (hence the safety factor) will be attained at each temperature level down to the minimum without a severe loss of nitrification.

The SRT data obtained from the above study can be related to parallel observations of F/M and volumetric loading (Section 5.3.5) and thus arrive at aeration volume requirements. This data will also produce secondary clarifier solids loading values ( $\text{lbs/ft}^2/\text{hr}$ ) for design purposes, by using maximum aeration MLSS and expected wastewater flow from the aforementioned study.

## 8.2 Existing Facilities

The following questions should be answered prior to a plant treatability study for upgrading an existing system for nitrification:

- (a) To what extent was nitrification achieved in the past?
- (b) Under what process and/or climatic conditions was nitrification achieved?
- (c) Have there been any changes in the raw wastewater flow-rates and constituents?
- (d) Is denitrification likely to be included in the future?

In response to questions (a) and (b) a survey may reveal that, for example, a facility achieved a target effluent ammonia -N of 1 mg/l during the summer months. Conversely effluent ammonia -N levels rose to a mean of 10 mg/l during the winter months under similar process SRT, F/M ratio and wastewater content.

This data would therefore indicate that conditions are only inadequate for nitrification at lower wastewater temperatures. Air supply may probably be adequate, assuming no further increase in loading,

since ammonia oxidation was maintained during summertime high oxygen demand period.

A treatability study format arising out of the above observations would be relatively simple. A section of the facility could be isolated and nitrification efficiency monitored during October to March utilizing higher than previously observed SRT (refer to Appendix 13.4.3). It should be noted that if a treatability evaluation is commenced in mid-winter, there may be considerable delay in obtaining an active aeration basin bacterial culture due to previous low temperature. Other meaningful information besides SRT can be gained from a full-scale evaluation. The performance of the aeration at higher solids concentration, in regard to mixing and oxygen transfer would be invaluable in deciding requirements in that area. Also, the efficiency of the secondary clarifier under the increased solids loading would be tested as well.

Along with a semi-full-scale evaluation, a lab or pilot-plant study could be incorporated and run at a different wastewater temperature and SRT. This could shorten the required study period and add to the data. If a WPCP produces little or no nitrification year-round despite adequate conditions in terms of requirements stated in Section 5, the influence of toxic materials or erratic flows can be suspected. Then in relation to question (c), a knowledge of upstream influxes of toxic wastes and flow extremes is of prime importance. It makes little sense to design an activated sludge system based on predetermined limits on SRT etc. if these conditions are only met at infrequent or random manner.

Notable examples of upstream hydraulics effecting nitrification systems are those of Newmarket and Stratford, Ontario WPCP's. Operational data reveal that nitrification is impeded in these plants during and



following periods of stormwater infiltration. This is caused by a loss of process solids (lowering SRT) during higher-than-design flow periods.

In cases like this, attention should be focused upon correcting sources of infiltration and/or implementing flow equalization prior to primary treatment before considering secondary expansion.

The introduction of toxic wastes by industry requires a large SRT safety factor and thus increased size and cost for the facility (Section 5.3.3). Considerable saving in cost for biological treatment can be attained if this waste can be treated at source via a more efficient physical and/or chemical means (53). If for some reason this option is not feasible, a treatability study comparing a two-sludge to a single-sludge nitrification system is recommended.

If the answer to question (d) is positive and denitrification may be a future requirement, some provision for this step should be included in the plant expansion. An area for adaptation to (slow mixing anoxic mode) denitrification, can be reserved by placing air turbine equipment in a section of the aeration to be operated in an aeration or mixing only mode. Another option is the adaptation of surface aerators to mixers such as was done for the study at Newmarket (12).

### 8.3 Nitrification/P-Removal

In many areas phosphorus removal will also be required and therefore should be part of the treatability study. Nitrifier growth rates are effected by aeration pH (Section 5.3.1) which in turn influences SRT. Since some P-removal chemicals remove alkalinity, particular

attention should be focused on this area during a treatability schedule. If the wastewater is low in alkalinity, then probably the addition of an alkaline substance will be needed and should be evaluated in the study.

#### 8.4 Effluent TOD and Filtration

If a facility is designed on the basis of allowable TOD discharge, then the determination of the effluent R factor (as outlined in Section 6.4) should be included in a treatability evaluation. This factor can then be applied to the TOD calculation along with TKN and  $BOD_5$  results and an effluent quality can be predicted.

In municipal plants a secondary effluent suspended solids concentration of 15 mg/l will normally represent between 2 and 3 mg/l of organic nitrogen; which is of course represented in the TKN measurement. Taking this into consideration, it is possible that at an existing site where complete nitrification occurs, only effluent filtration may be required. A tertiary process such as this alone may reduce effluent suspended solids (organic-N) and  $BOD_5$  down to an acceptable TOD level.

#### 8.5 Future Community

If the plans for a future community call for a nitrified WPCP effluent discharge, probably the best approach would be to recommend an extended aeration facility for good year-round ammonia oxidation. An alternate plan would be to build a primary plant with flow equalization and design a secondary stage at a later date based on a treatability study.

## 9. PROCESS ECONOMICS

A detailed analyses of unit costs for nitrification will not be attempted as this would require a report in itself; but some important factors should be considered prior to designing and/or expanding secondary facilities for nitrification. As stated in Section 5.2, by enlarging the primary facility capacity, a decreased organic loading to secondary facility will result. Therefore, in plants where ammonia oxidation occurs in the process to some degree and only a marginal improvement is required, the expansion of the primary clarification facilities may often be a more economical choice than an enlargement of the aeration unit.

Two other alternatives to secondary capacity expansion would be to consider lime addition to the raw sewage such as studied at the Newmarket WPCP (9) or placement of flow equalization prior to primary clarification (Section 5.2).

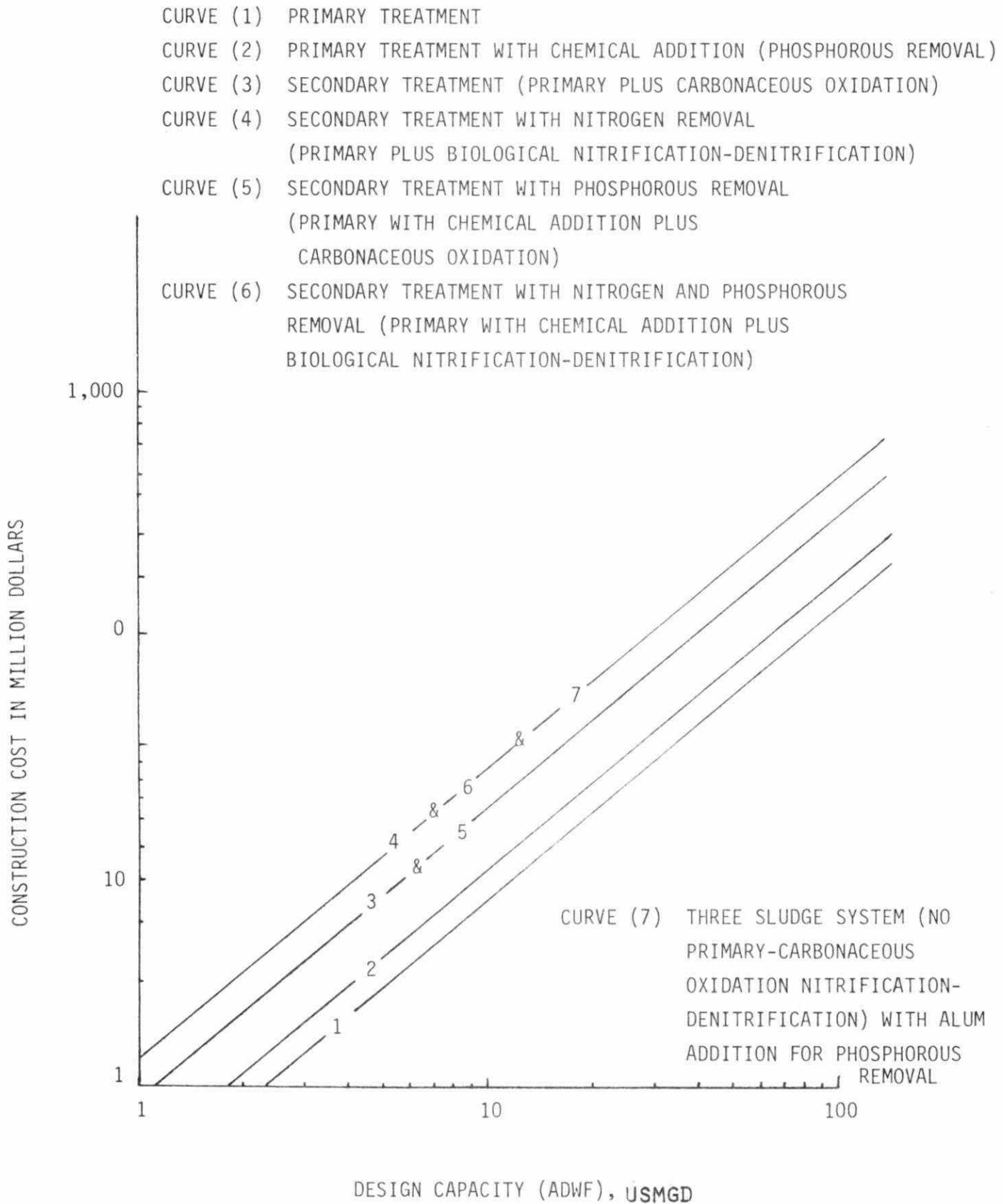
The information shown in Table 8 taken from recent costing curves (Fig. 12 (a and b)) shows that construction costs based on a 10 mgpd plant for a primary unit are comparable to those of a secondary facility. The primary unit includes facilities such as screening, flow measurement and grit removal, therefore, the primary clarifier cost would only be a fraction of the costs for the complete primary unit as given in Table 12. Moreover, chemical addition for P-removal represents only 17% of primary unit cost.

Figures on operating costs indicate a slight advantage of secondary over primary facilities, but since the secondary treatment costs quoted are for carbonaceous removal only, it can be expected that a higher cost would be incurred if nitrogenous oxidation were included (see Figure 12).

FIGURE 12

COST FOR WASTEWATER TREATMENT

(a) Construction



(b) Operation

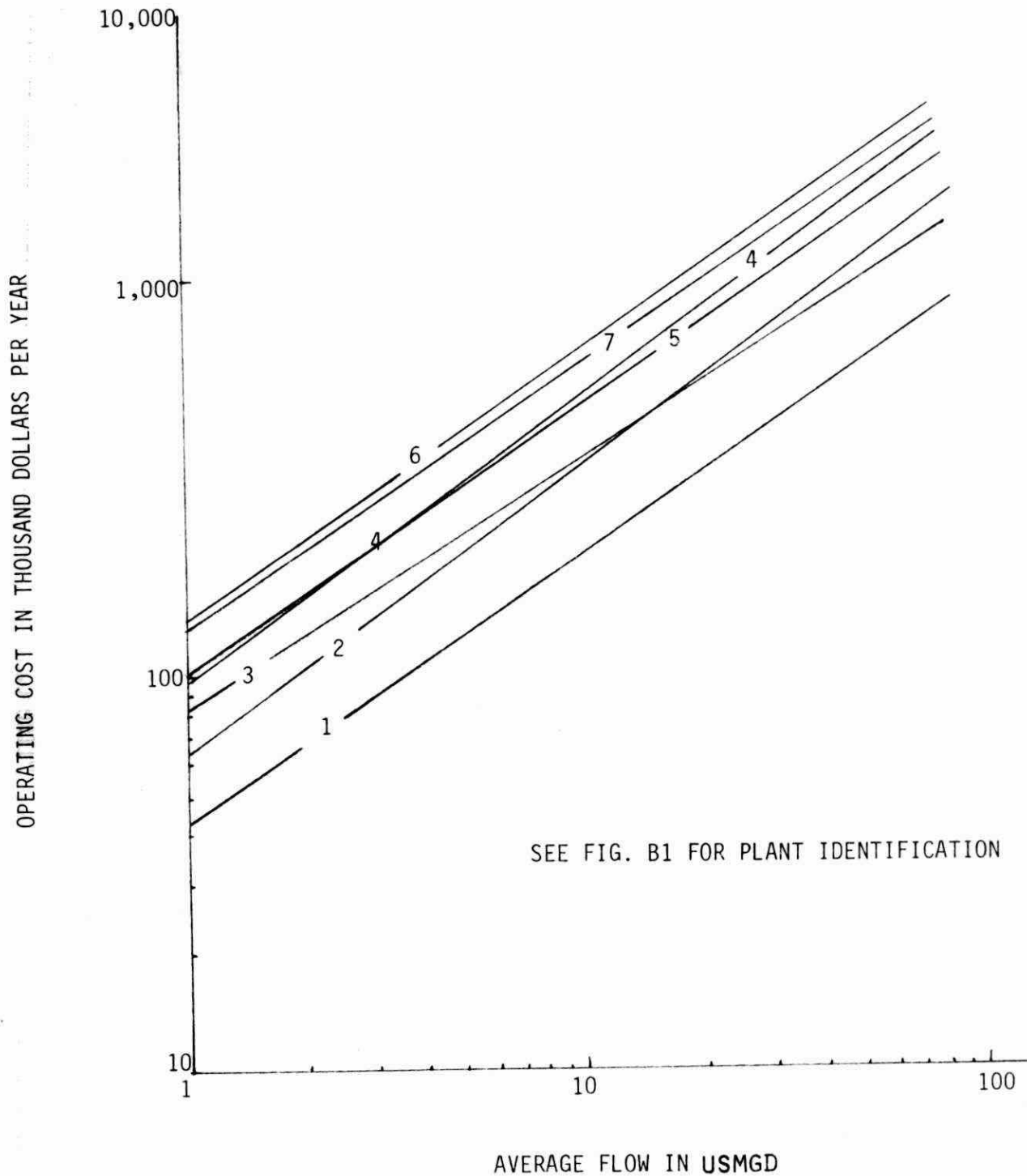


TABLE 9

COST OF WASTEWATER TREATMENT  
BASED ON A 10 USMGPD PLANT (54)

Construction Costs - \$ x 1,000,000

1. Primary Unit	= 3.5
2. Primary plus Chemical-P	= 4.2
3. Primary plus Secondary (Carb)	= 6.5
4. Secondary only 3-1	= 3.0

Operating Costs - Thousand \$/year

1. Primary Unit	= 200
2. Primary plus Chemical-P	= 350
3. Primary plus Secondary (Carb)	= 360
4. Secondary only	= 160

Flow Equalization Facilities (55) - \$ x 1,000

<u>Earthen Basin</u>		<u>Concrete Basin</u>	
With Pumping	Without Pumping	With Pumping	Without Pumping
318	134	779	595

From these observations it can be realized that enlargement of the primary clarification and/or chemical addition prior to secondary treatment is a viable alternative. As outlined in Section 13.1, equalization prior to primary treatment can also increase the efficiency of the primary and reduce loading surges to the aeration basin.

Judging from the values given by the EPA (55), flow equalization would cost \$779,000 to treat 10 U.S. million gallons per day based on a tank volume of 2.8 U.S. mg. Consequently, this system may also offer an economical choice (depending on land and operating costs) compared to the secondary expansion costs (11) for good nitrification.

The ultimate decision as to what area of an existing facility should be enlarged for nitrification will depend on locality, availability of land, existing equipment and labour costs. Consequently, the aforementioned data can only be used as a yardstick.

## 10. DISCUSSION

This report has presented the status of the current knowledge of the nitrification process as applied to activated sludge plants.

The following facts need to be re-emphasized:

(i) It is extremely important to do complete wastewater analyses and treatability studies in order to make realistic recommendations on the design of a system.

(ii) The design data contained in this report are only suggested guidelines and should be treated as such. Only individual plant surveys and studies can be expected to yield the required design information.

(iii) As more full-scale studies of the nitrification process take place, it is expected that additional information regarding process parameters may evolve. The current experience of the Ministry of the Environment is based on the investigations carried out with the single-stage, combined sludge design and is therefore limited in scope.

(iv) Although not mentioned earlier in this report, considerable attention has been paid recently to those designs of nitrification units which may be used as an "add-on" tertiary system at existing activated sludge plants treating carbonaceous BOD. Some of these units include trickling filters with recycle, rotating biological discs or drums, etc. It is hoped that some of these may be evaluated and reported on during the coming year by the Wastewater Treatment Section.



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## 12. APPENDIX

### 12.1 Flow Equalization

As commented earlier in Section 5.1.1, the principle of flow equalization is attractive from the standpoint of buffering not only the hydraulic changes but the diurnal fluctuations in waste concentrations. Since the nitrification process efficiency is more sensitive to environmental changes, the need for equalization becomes more apparent in an effort to maintain a stable process. The following summary is an extract of information derived from the US E.P.A. design manual (18) for Upgrading Existing Wastewater Treatment Plants. For additional facts on equalization techniques the reader is directed to reference numbers 56, and 57.

#### 12.1.1 In-Plant Equalization

In-plant equalization basins should normally be located between the upstream screening and grit removal and the primary clarifier, or in the case of extended aeration, prior to the aeration basin. The basins may be placed as either in-line or side-line units; the in-line units offering better equalization opportunity. A description of these two arrangements is shown in Figure 13.

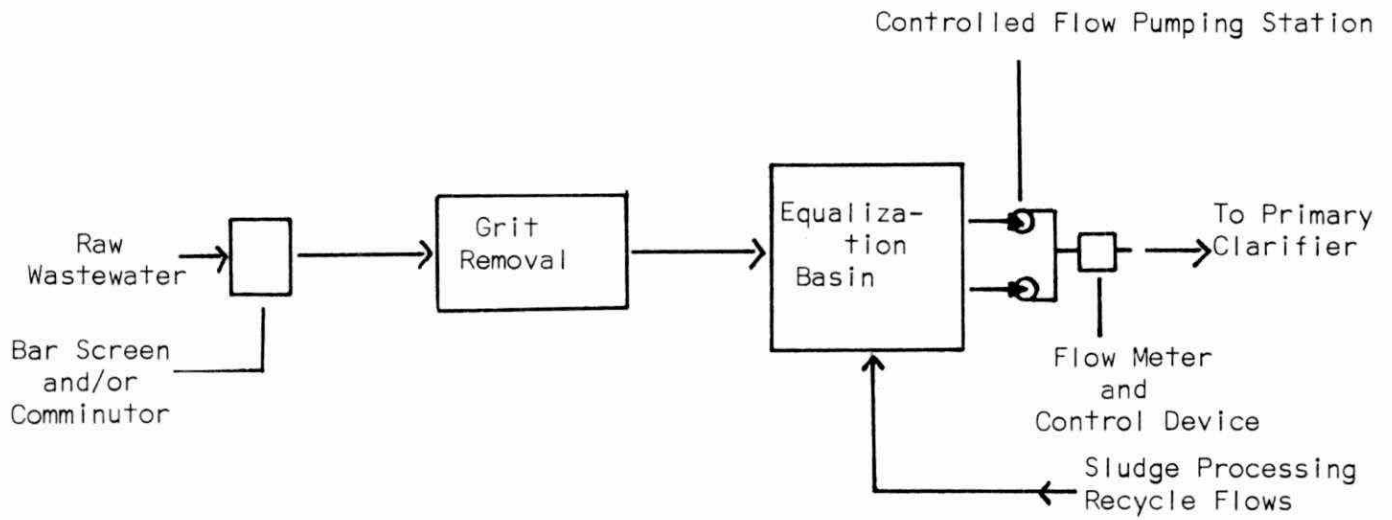
If a particular facility experiences drastic flow increases due to stormwater infiltration, a modular or compartmentalized designed basin is recommended. In this arrangement only part of the equalization unit need be used during dry weather flow periods.

#### 12.1.2 Benefits of Equalization

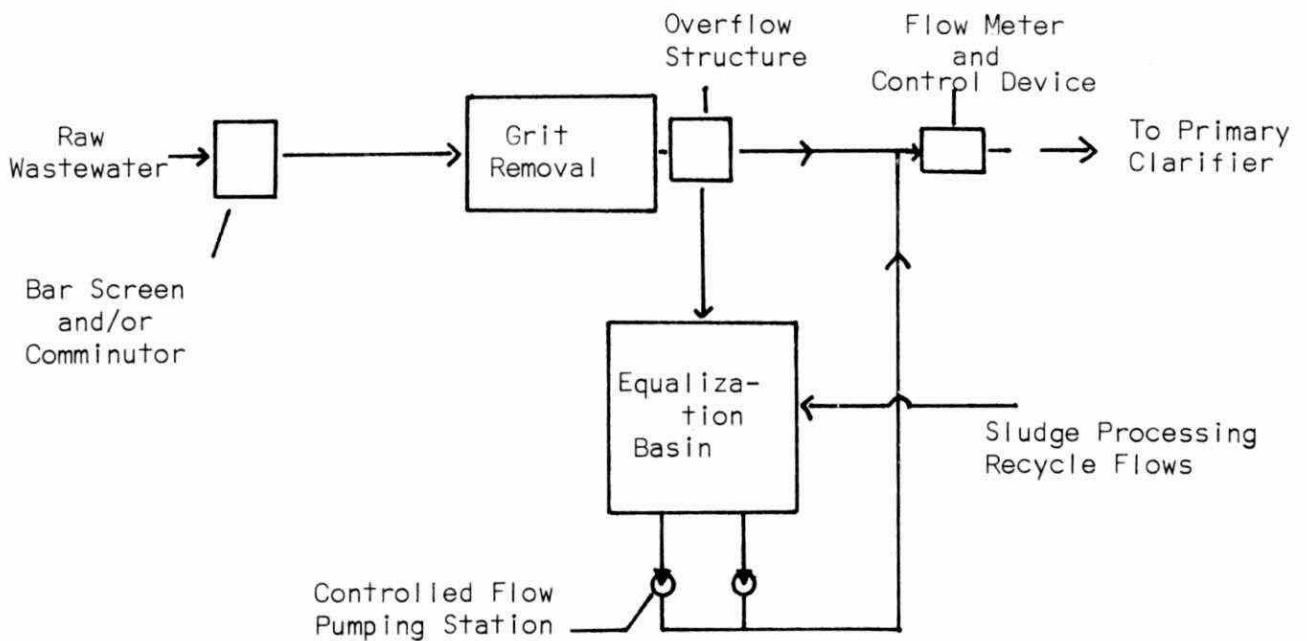
The benefits of flow equalization are first realized in the performance of the primary clarification stage. Studies in Newark, New York showed a 24% improvement in suspended solids reduction across the primary with equalization applied (see Table 10 below).

FIGURE 13

SCHEMATIC FLOW DIAGRAMS OF EQUALIZATION FACILITIES  
(U.S.E.P.A.)(55)



IN-LINE EQUALIZATION



SIDE-LINE EQUALIZATION

TABLE 10

EFFECT OF FLOW EQUALIZATION ON PRIMARY SETTLING

NEWARK, NEW YORK (LA GRECA AND KEENAN) (57)

	<u>Normal Flow</u>	<u>Equalized Flow</u>
Primary Influent SS (mg/l)	136	128
Primary Effluent SS (mg/l)	105	68
SS Removal (%)	23	47

Further benefits are produced in the biological process such as the levelling of waste pH fluctuations before entering to the aeration basin. Air requirements are based on peak carbonaceous and nitrogenous load, thus a levelling of these peaks will decrease needed aeration capacity. Since more uniform growth will occur, the organic loading ratio and process SRT will be more controllable. The final clarification stage will operate more efficiently as fluctuations in flow rate and sludge blanket levels will be minimized (55, 57).

12.1.3 Design Considerations

Normally the design volume of an in-plant, dry weather flow equalization basin is 10 to 20% of the expected average dry weather flow. When designing for wet weather as well, a detailed investigation of waste flow pattern should be conducted.

It is important that complete mixing is provided preferably by air or combination air-and-mixer systems to prevent septic conditions. Mixing requirements for a municipal wastewater having a suspended solids concentration of 200 mg/l, range from 0.02 to 0.04 HP/1,000 gallons US (180 m<sup>3</sup>) of sewage. To maintain aerobic conditions in the equalization basin, air should be supplied at 1.3 to 2.0 cfm/1,000 US gal (55) (0.010 to 0.014 m<sup>3</sup>/min/m<sup>3</sup>).

The equalization basin should be square in shape with the inlet close to basin mixers or aerators to prevent short-circuiting. A sloped bottom is preferred with a drain for cleaning purposes and a bypass channel to the primary clarifiers for use when the basin is shut down.

Pumping may be required to feed the equalization basin if the ground elevation is not conducive to gravity feed. Outlet pumping should be provided by a combination of constant and variable speed pumps.

## 12.2 Toxic Materials

As discussed in the preceeding paragraphs, the role of toxic substances has received considerable attention by investigators of nitrification processes as the presence of these materials can have a major influence over the size, design configuration, and ultimately on the cost of a facility. Generally, the types of wastes which may be toxic to nitrification bacteria can be classified into three groups - metallic compounds, organic compounds and oxidizing compounds.

### 12.2.1 Metallic Compounds

Studies by Loveless and Painter (58) with an isolated culture of *Nitrosomonas* from activated sludge have shown that depending on the metal concentration, the bacterial growth rate may be stimulated or inhibited. For example, soluble copper concentrations below 0.03 mg/l often increased growth whereas a range of 0.05-0.56 produced an inhibition. Furthermore, the effect of pH on the *Nitrosomonas* was dependent on the metal ion concentration. The toxicity of copper was found to be non-existent up to 0.48 mg/l concentration in the presence of EDTA (Ethylenediamine tetra - acetic acid) (See Table 11).

Calcium and magnesium also gave a stimulation of growth rate at low concentrations but the latter showed an inhibition above 50 mg/l. Sodium was found toxic at more than 0.2% concentration in chloride or sulphate form.

Similar investigations by Skinner and Walker (59) with pure cultures have shown that above 0.25 mg/l, chromium, nickel and copper were toxic to nitrification bacteria. Zinc, manganese and cobalt were not found toxic at 1 mg/l. The accumulation of copper in the activated sludge due to successive additions caused complete suppression of nitrification. A loss in nitrification has also been noted by the author with the addition of copper sulphate for filamentous bacteria control (60).

TABLE 11  
SUMMARY OF THE EFFECT OF VARIOUS METALS ON GROWTH  
OF NITROSOMONAS  
(Loveless and Painter, 1968) (58)

Element	Range Tested	Concentration at which effect observed (mg/l)	
		<u>Stimulation</u>	<u>Inhibition</u>
Cu	0.005-0.56	0.005-0.03	0.05-0.56
Cu*	0.005-0.48	none	none
Ca	0.5 -20	none	none
Ca*	0.5 -20	0.5 -10	none
Mg	12.5 -100	12.5 -50	50-100
Zn	0.005-0.5	none	0.08-0.5
Co	0.005-0.5	none	0.08-0.5
Al	0.005-1.0	none	none
Sr	0.005-0.5	none	none
Pb	0.005-0.05	none	none
B	0.005-0.05	none	none

\* In presence of 5 mg/l EDTA (Ethylenediaminetetra-acetic acid)

Information from pilot and full-scale activated sludge process evaluations have indicated that the levels of metals required for nitrification inhibition may be somewhat higher than the values quoted here, suggesting that bacteria may acclimate to continuous low levels of metallic compounds under actual process operation. Rimer and Woodward (15) found that copper in a soluble form at 5 mg/l in the primary effluent inhibited nitrification in a two-stage separate sludge process. Beckman et al (2) reported no toxicity to nitrifiers with copper or hexavalent chromium up to 0.5 mg/l; nickel and zinc did retard nitrification at levels above 1.5 mg/l.

Taking into consideration the present available information, a safe maximum level for individual heavy metals tolerable in raw sewage or primary effluent might be 1 mg/l assuming that some acclimation of the bacteria to the metals might take place. Particular importance must be placed on the solubility of the metal compound and therefore its availability and hence its toxicity to the bacteria.

#### 12.2.2 Organic Compounds

Extensive investigations by Downing et al (11) in 1974 brought forth much needed information as to the effect of organic compounds on nitrification reactions. Experiments were conducted on unadapted activated sludge taken from a local WPCP and batch tests were run in which a variety of organic compounds were tested for their effects on nitrification. The relative concentrations of these compounds to produce 75% inhibition of nitrification were catalogued and are listed in Table 12. Generally, the compounds found to be the strongest inhibitors were those compounds in which sulphur and nitrogen were attached to the same carbon atom, such as thiourea and thioacetamide at concentrations of 0.075 and 0.14 mg/l respectively. The relative toxicity of these organic compounds is illustrated in Figure 14 derived from the work of Downing et al (11). Compounds used as accelerators in the rubber industry and as fungicides

TABLE 12  
EFFECT OF ORGANIC COMPOUNDS  
ON NITRIFICATION (DOWNING et al) (6)

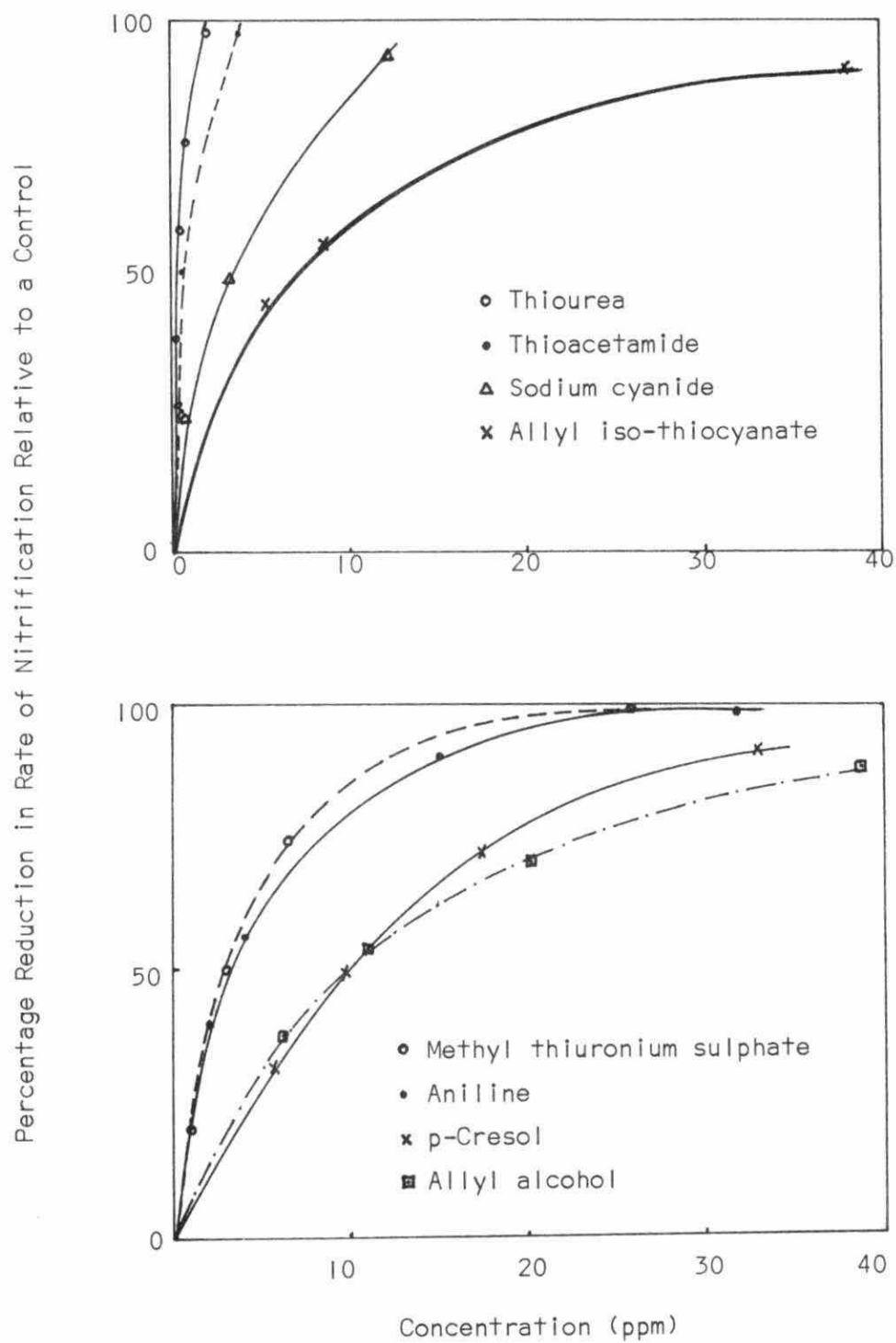
Compound	Concentration Necessary to Produce 75% Inhibition of Nitrification mg/l
Thiourea	0.075
Thioacetamide	0.14
Thiosemicarbazide	0.18
Methyl isothiocyanate	0.8
Allyl isothiocyanate	1.9
Dithio-oxamide	1.1
Potassium thiocyanate	300
Sodium methyl dithiocarbamate	0.9
Sodium dimethyl dithiocarbamate	13.6
Dimethyl ammonium dimethyl dithiocarbamate	19.3
Sodium cyclopentamethylene dithiocarbamate	23
Diperidinium cyclopentamethylene dithiocarbonate	57
Methyl thiuronium sulphate	6.5
Benzyl thiuronium chloride	49
Tetramethyl thiuram monosulphide	~ 50
Tetramethyl thiuram disulphide	~ 30
Phenol	5.6
o-cresol	12.8
m-cresol	11.4
p-cresol	16.5
Dichlorophenol	> 50
2-4 dinitrophenol	> 150
Allyl alcohol	19.5
Allyl chloride	180
Diallyl ether	100
Sodium cyanide	0.65
Aniline	7.7



TABLE 12 (continued)

	Concentration Necessary to Produce 75% Inhibition of Nitrification mg/l
Dimethyl-paranitrosoaniline	19
Guanidine carbonate	16.5
Skatole	7.0
Strychnine hydrochloride	175
2-chloro-6 trichloromethyl-pyridine	100
Ethyl urethane	> 250
Ethyl diamine tetra-acetic acid	350
Hydrazine	58
Methylene blue	> 100
Carbon disulphide	35
Acetone	840
8-hydroxyquinoline	73
Streptomycin	> 400

FIGURE 14  
RELATIVE TOXICITY OF ORGANIC COMPOUNDS TO NITRIFICATION RATE  
(Downing) (6)



such as dithiocarbonate and thiuran sulphide and cyanide compounds showed toxicity at concentrations below 2 mg/l.

### 12.2.3 Oxidizing Compounds

The use of oxidizing compounds such as chlorine and hydrogen peroxide has been common practice for the treatment of bulking sludges for a number of years (61)(62). The process is normally disrupted in the first two days of application but the  $BOD_5$  removal process recovers within two weeks. Studies by the author (32) have shown that under these conditions, nitrification is only partially affected and recovers along with carbonaceous oxidation.

Difficulties arise when a prolonged treatment is required, such as the dosing of a sludge having a sludge volume index (SVI) of more than 300 ml/g with chlorine over a period of one week. The problem is compounded if the addition of oxidizing chemicals coincides with low sewage temperatures and a resultant low bacterial growth rate. These compounds will cause destruction of large numbers of nitrifying bacteria along with the filamentous organisms.

Experience with full-scale nitrification and nitrification-denitrification processes at the Newmarket WPCP (12) showed a marked decrease in nitrification efficiency after repeated application of chlorine additions, and caused such a disruption to the process that deflocculation occurred with a resultant loss of solids to the final effluent. A marked decrease in nitrification efficiency occurred during the winter months and was not recovered fully until April. During the recovery phase, levels of nitrite rose above 1 mg/l as nitrification was regained; this being the only study period where nitrite levels were abnormally high. Recent studies being conducted at the O.E.F. show the appearance of nitrite under process deterioration, consequently, this compound is thought to indicate a disturbance to the nitrification system.

The use of hydrogen peroxide in the activated sludge process has been recommended commercially for the removal of hydrogen sulphide, for the prevention of denitrification in secondary clarifier sediments, for the treatment of bulking sludges and as a supplementary oxygen source for biological waste treatment (61) (62). Large dosages are usually required for the control of bulking activated sludges and a reduction in SVI may require as much as 500 mg/l  $H_2O_2$  to the recycle flow.

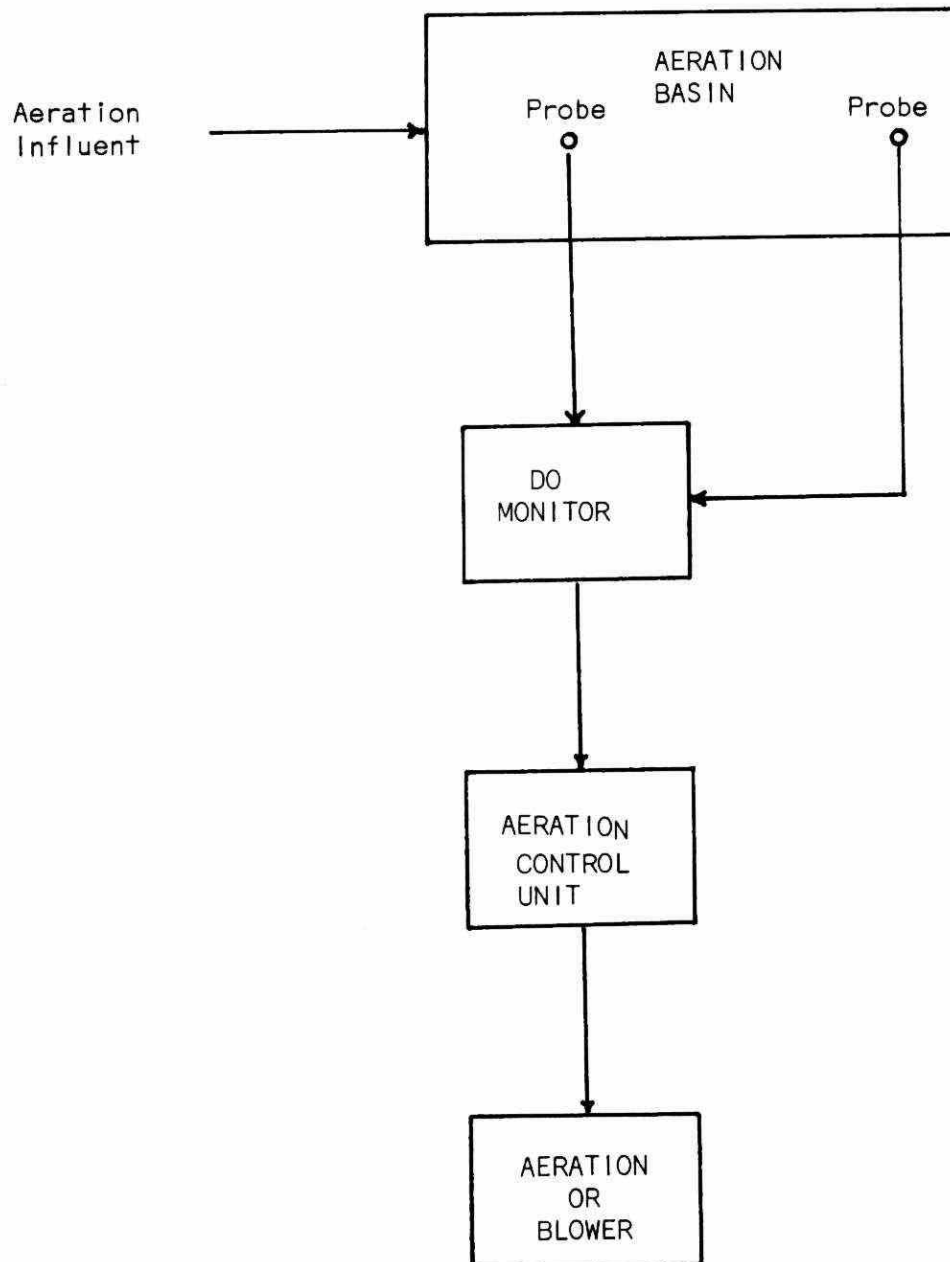
Hydrogen peroxide was applied to the OEF nitrification-denitrification process to control a bulking sludge problem, and short duration additions of between 25 and 100 mg/l caused a drop in nitrification efficiency from 95% to 30% over a one month evaluation period (32). It was concluded that further work, in which a lower dosage with multi-point addition with emphasis on monitoring of nitrification reactions is required.

### 12.3 Automatic Control of Oxygen for Aeration Basin

The automatic control of aeration has been investigated by Bernard and Lauboutin (63) and Jones and Briggs (64) and it would seem that most facilities would achieve some power savings with automation. The automated arrangement calls for a system of DO sensors placed in the aeration mixed liquor at locations considered to be critical areas for a minimum oxygen concentration; one or two probes may be sufficient in a well mixed basin. These probes relay the mixed liquor DO concentrations to a controller circuit which can be set to maintain DO levels within required limits by varying the air supply system output, (see Figure 15).

The signal from an automatic DO feedback system can be directed to either an air blower, air turbine or mechanical aeration arrangement. Caution must be exercised, when converting old or designing new systems, that adequate mixing is maintained in the aeration basin at low air delivery periods. For mechanical aerators with multiple or

FIGURE 15  
AUTOMATIC AERATION CONTROL SYSTEM



two-speed drives, an extended shaft with a propeller may be required to prevent sludge deposition in the basin at low speeds. Combinations of constant and variable speed blowers are more economical electrically than an all variable system. The submerged turbine system is well suited for an automated process as mixing is constant and the air delivery is varied.

During the winter months with low liquid temperatures, oxygen transfer is normally improved in an aeration basin. Mixed liquor DO concentrations particularly during low load periods could be well above what is required to maintain the process, consequently, needless energy is being expended. From this standpoint alone, the advantages of an automated or a more controllable aeration system is realized.

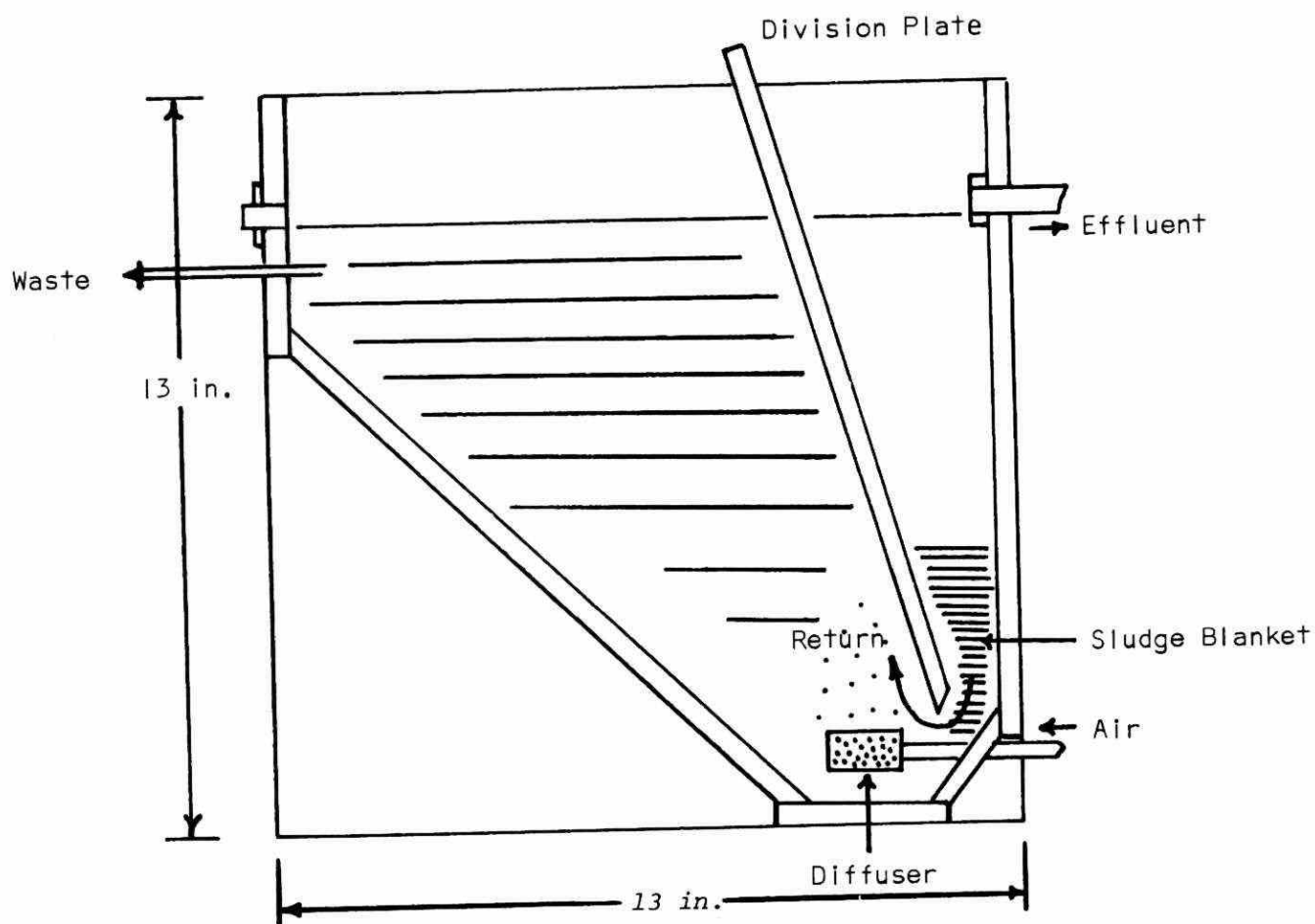
#### 12.4 Treatability Studies

##### 12.4.1 Laboratory Test Units

Laboratory size treatability units, such as illustrated in Figure 16 are normally used when the influence of toxic substances on the activated sludge mass is to be studied. Constructed of plexiglass, these miniature activated sludge plants are easy and inexpensive to make and several can be operated in parallel without too much operator attention. The unit is a combination of an aeration and clarifier section. Primary clarifier effluent or settled raw sewage is introduced via a small pulsating or continuous flow pump. Aeration is provided by an oilless lab compressor and an aquarium size stone diffuser.

The turbulence and flow patterns generated by the aerator and spacing of the adjustable division plate determines the rate of recycle of settled solids from the clarifier section. Unfortunately, the rate of recycle sludge, which is an important parameter for nitrification evaluations, cannot be measured. Wasting of sludge from the unit is usually done from the aeration section via a pump and adjustable timer.

FIGURE 16  
LABORATORY TEST UNIT FOR TREATABILITY STUDIES  
(Johnson and Schroepfer)(65)



DETAILS OF NITRIFICATION REACTOR UNIT

In the author's opinion laboratory units that employ separate aeration and clarifier sections have been used with only limited success. Difficulties occur in designing a clarifier sludge withdrawal system which will adequately gather and transport settled solids back to the aeration section. Clarifier sludge denitrification is often a problem with these units if the nitrified sludge blanket is allowed to become anoxic.

#### 12.4.2 Pilot Plant Units

Due to their sizes, pilot plant test units more closely resemble the full-scale or semi/full-scale unit but with the distinct advantage of allowing closer control over parameters such as flow, temperature and wasting, etc. Clarifier sludge withdrawal is not normally a problem, as with lab units, because an adequate sludge scraper can be installed. Furthermore, the recycle stream piping is large enough to transport sludge without clogging. A diagram of a 260 gal/day ( $1.2 \text{ m}^3/\text{day}$ ) single-sludge unit used by Hall in 1974 (29) is illustrated in Figure 17.

A unit similar in nature to this has been used by Sutton et al (4) at the Canada Centre for Inland Waters (CCIW) with added flexibility for various process modes of operation.

The relative size of the pilot plant equipment usually facilitates longer periods of operation without mechanical breakdowns and resultant process disruption.

To facilitate portability, a pilot unit can be mounted on a covered truck or trailer with a step-down transformer and associated power outlets for pumps and samplers, etc.

#### 12.4.3 Semi/Full-Scale Unit

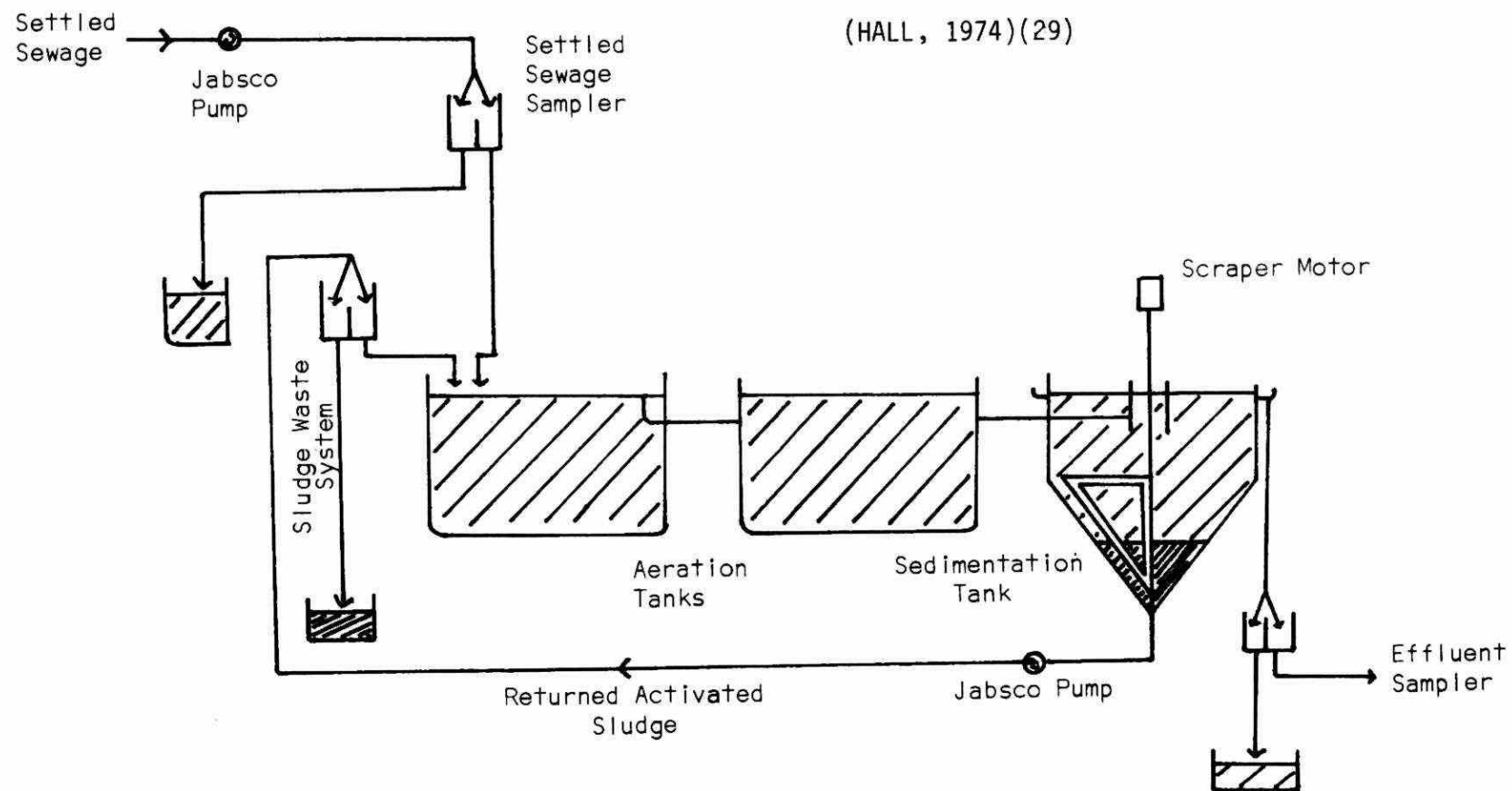
When time is not a major factor, a semi/full-scale unit treatability study can be most productive because of its similarity to the eventual process conditions. Usually, a section of an existing plant



FIGURE 17

SCHEMATIC DIAGRAM OF PILOT-SCALE ACTIVATED SLUDGE PLANT

(HALL, 1974)(29)



is isolated and operated under conditions conducive to process nitrification and the performance of the units is studied under varying climatic and waste conditions. These studies can last for many months and a great deal of operating experience can be gained using full-scale equipment for optimizing the nitrification efficiency.

Before choosing the semi/full-scale treatability evaluation over a lab or pilot unit, one must be sure that a plant section can be isolated without expensive alterations, aeration capacity is adequate, recycle and waste sludge flow is measureable and some means of waste flow division and monitoring is available separate from the remaining plant facility.

### 12.5 Disinfecting Nitrified Effluents with Chlorine

There are several differences between the chemical and bacteriological characteristics of nitrified effluents (1-5 mg/l  $\text{NH}_3$ ) and typical secondary effluent (5-15 mg/l  $\text{NH}_3$ ) that can significantly affect the operation and control of the disinfection process. The first three conditions outlined below occur when the biological treatment, including secondary clarification, is working well, the last, when it operates poorly.

1. There is a smaller range and lower median count of those indicator organism groups (Total Coliform, Fecal Coliform) commonly used to assess the efficiency of disinfection. Both factors enhance the chances of obtaining good disinfection efficiency.

2. Chlorine demand tends to be more stable, i.e. extreme values fall within narrower limits as milligrams/litre. This factor also tends to enhance the chances of obtaining better disinfection, since a desired level of chlorine residual can be maintained more easily when controlling the addition of chlorine by simple methods such as flow pacing.

3. If the nitrification process should proceed virtually to completion ( $\text{NH}_3$ -N levels 0.3 mg/l or less) due to low organic and/or nitrogenous loadings, then there is the possibility that breakpoint chlorination may occur. This is undesirable in a conventionally designed chlorine disinfection system, since chlorine which has volatilized to the atmosphere is not available for disinfection. The plant operator will become aware that breakpoint chlorination is occurring, by a significant increase in chlorine demand in an otherwise normal effluent (high nitrate and low ammonia-N levels) and accompanying decline in end-of-contact residual.

It should be noted that designing the system to achieve breakpoint chlorination would generally involve an addition of at least 14 mg/l of chlorine, based on removing 1 mg/l of ammonia, and satisfying a chlorine demand of 4-5 mg/l. Only above this level of dosage could the superior disinfecting power of free chlorine ( $\text{HOCl}$ ) become available.

A minimum of 1 mg/l of ammonia in the effluent is usually sufficient to avoid breakpoint chlorination.

4. Nitrite in the wastewater is readily oxidized to nitrate by chlorine. When high levels of nitrite are present in the effluent (3 mg/l) there will be significant oxidation upon addition of chlorine to the wastewater. Following a period of plant upset, high nitrite levels could persist for some time, and in extreme cases, for several days.

When nitrite levels are present, then it may become difficult to obtain a combined chlorine residual. The best approach to solving the problem is to concentrate on restoring the biological plant operation to achieve normal nitrification levels. A poor alternative is to dose very large amounts of chlorine to the effluent.

#### 12.5.1 Analysis of Total Combined Chlorine Residual in Nitrified Effluent

Whilst the orthotolidine method is subject to many interferences in typical wastewater, small quantities of nitrite will be present regularly in undisinfected nitrified wastewater. The nitrite, not normally present in non-nitrified wastewater, will cause interference with the orthotolidine test and result in higher results for total combined chlorine residual than the true value.

The amperometric method, using phenyl-arsene-oxide as the titrant, is the analytical method of choice.